SEQUENTIAL NITRIFICATION/DENITRIFICATION

IN

SUBSURFACE FLOW CONSTRUCTED WETLANDS

A LITERATURE REVIEW

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BY

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I. INTRODUCTION

A. Introduction

A need exists for the application of a low cost, low maintenance, low technology municipal wastewater treatment option for small rural communities. The construction, operation, and maintenance of a community wide managed wastewater facility is a major and difficult undertaking for most small (<1000 pop.) communities. Small communities face the problems of stringent discharge requirements, high per capita costs, limited finances, and limited operations and maintenance budgets (Tchobanoglous and Burton 1991).

Over the last 50 years the responsibility for ensuring water quality has gone full circle. Responsibility for water quality has shifted from State governments to the Federal government, and is now moving back to the States. The Federal Water Pollution Control Act (FWPA) of 1948 was one of the first national legislative efforts to deal with water quality problems (Smith 1987). The FWPA goal was improved water quality, but enforcement and funding was primarily a state responsibility. Little progress was made on a national scale, and many Federally legislative initiatives were passed culminating in the Water Pollution Control Act Amendments (WPCA) of 1972 (PL 92-500) and the subsequent Clean Water Act Amendments (CWA) of 1977 (PL 95-217).

The WPCA and CWA mandated water pollution control for both municipal and industrial point source discharges, initiated a Federal program for non-point source water pollution control, and required control of toxic pollutants (Smith 1987). The most important part of these acts were that they provided Federal money to help state and local communities fund municipal wastewater treatment facilities.

Responsibility for water quality is now shifting back to the states with the passage of the Water Quality Act Amendment (to the CWA) in 1987. The Federal grants program for municipal wastewater treatment has been phased out and replaced with a State Revolving Loan Program funded through 1994 (Smith 1987).

Federal money is becoming difficult to obtain for small communities to fund and operate conventional wastewater treatment systems. As stated earlier, a need exists for a low cost treatment system that is applicable to the rural environment. Subsurface flow constructed wetlands (SFCW) may help small communities solve their wastewater problems. Reed and Brown (1992) report an average cost for construction of SF systems to be about \$0.62/gallon (flow), and Jones reports (1992) for operation and maintenance about \$0.18/1000 gallons (flow). These low costs, coupled with the reliable performance of SFCW for BOD and TSS removal, make the consideration of these systems a good choice.

Even though there is currently no consensus on the design of SFCW (Reed and Brown 1992), the ability of constructed wetlands to meet municipal wastewater requirements for BOD and TSS is well documented. Nitrogen removal appears from the existing performance data to be one of the primary problems with these systems. The negative effects of excessive levels of nitrogen on the aquatic environment include eutrophication of receiving waters and the increased risk of methemoglobinemia in human infants where elevated levels of nitrate (NO_3^-) or nitrite (NO_2^-) nitrogen are present in drinking water supplies (Shearer et al. 1972). The performance of constructed wetlands for nitrogen removal, at best, can be rated poor to fair (Gersberg et al. 1986, Watson et al. 1988, Chalk et al.1989, Brix et al. 1989, Choate 1989, Schierup et al. 1990, Conley et al. 1991, and Watson et al. 1992). As a result of the negative effects of excessive nitrogen on the environment and the problems with constructed wetlands in consistently removing nitrogen to within acceptable levels, this report will be directed towards the sequential nitrification/denitrification process.

B. Purpose

The purpose of this report is to determine the current state of SFCW technology and performance, with a primary focus on sequential nitrification/denitrification. This report has the following objectives:

1. To discuss the development of SFCW technology, review the variety of design approaches, and discuss the overall performance of these constructed wetlands;

2. To discuss nitrification/denitrification in SFCW and typical problems encountered;

3. To discuss emergent aquatic plants and oxygen transfer (the key to sequential nitrification/denitrification);

4. To review nitrification/denitrification performance data and develop recommendations on hydraulic loading, retention times, and nitrogen loading; and

5. To offer conclusions on the further applicability of SFCW technology and recommendations on further research needs.

C. Scope

The information for this report was obtained from a literature review consisting of: journal articles, conference proceedings, textbooks; EPA, TVA, and European publications; unpublished articles; and site visits to two constructed wetlands in Louisiana.

Objectives one, two, and three were met by reviewing the available current and past literature discussing SFCW and synthesizing the information into one concise report. Objective four was met by comparing available performance data from operating SFCW and developing recommendations based on their performance. Finally, conclusions and recommendations were based on the reviewed information and design guidelines presented.

II. CONSTRUCTED WETLANDS: AN OVERVIEW

A. Natural Wetlands

i. Introduction

Wetlands are areas that are periodically flooded with a frequency and depth sufficient to promote the growth of aquatic vegetation adapted to life in saturated soil conditions. Wetlands occur naturally throughout the world and function as transitional zones between purely aquatic ecosystems and uplands (WPCF 1990).

In 1979, the U.S. Fish and Wildlife Service developed a definition and classification system for natural wetlands. From their definition a wetland must have one or all of the following characteristics (Hammer and Bastian 1989):

- areas supporting predominantly hydrophytes (at least periodically);
- areas with predominantly undrained hydric soils (wet enough for long enough to produce anaerobic conditions that limit the types of plants that can grow); and/or
- areas with nonsoil substrata (such as rock or gravel) that are saturated or covered by shallow water at some time during the growing season.

Wetlands are classified into 5 major systems:

- Saltwater swamps - These are mangrove wetlands located on the southern coast of Florida and in Texas. Mangroves are among the very few woody plants that can tolerate saltwater conditions.
- Freshwater swamps - Freshwater swamps contain a variety of woody plants and water tolerant trees. Typical plants are as follows:

Southern	Northern
Bald Cyprus (Taxodium)	Alder (Alnus)
Tupelo Gum (Nyssa)	Black Ash (Fraxinus)
White Oak (Quercus)	Black Gum (Nyssa)
River Birch (Populus)	Northern White Cedar (Thuga)
	Tamerak (Larix)
	Red Maple (Aeir)
	Willow (Salix)

• Coastal salt marsh - Salt marshes are dominated by salt tolerant herbaceous plants, such as:

Cordgrass (Spartina)
Blackrush (Juncus)
Glasswort (Solicornia)

• Freshwater marshes - These marshes are dominated primarily by emergent hydrophytes such as:

Cattails (Typha)
Bulrush (Scirpus)
Reed (Phragmites)
Grasses
Sedges (Carex)

• Bogs - These types of wetlands are located primarily in the northeastern and north-central regions of the U.S. Bogs are dependent upon stable water levels and are characterized by acidic, low nutrient water and acid tolerant mosses. Typical plants include:

Sphagum moss
Cranberry (Vaccinium)
Tamarack
Black Spruce (Picea)
Leatherleaf (Chamadaphne)

ii. Wetlands: Natural Treatment Systems

Wetlands have many functions. They provide habitats for a huge and diverse amount of wildlife, stabilize shorelines, provide natural flood control by buffering peak rain fall events, function as groundwater recharge areas and as natural reservoirs by holding huge amounts of fresh water, and perhaps the most important and least understood function of wetlands is their ability to improve water quality (Hammer and Bastian 1989).

Wetlands have a tremendous assimilative capacity for pollutants and nutrients by a variety of naturally occurring physical, chemical, and biological activities. The quiescent water conditions are very conducive to the sedimentation of wastewater solids and wetland soils are effective natural filters. Chemical and biological reactions

breakdown complex compounds into simpler substances and some pollutants are physically or chemically immobilized and remain permanently unless disturbed. Natural wetlands support a large and diverse population of bacteria which grows on the roots and stems of aquatic plants and on the sediment. These bacteria are extremely effective in the removal of BOD_5 and in nitrification/denitrification. Also, through adsorption and assimilation wetlands remove nutrients for biomass production (Hammer and Bastian 1989, EPA 1988, Reed et al. 1985)

Wetland plants also possess a unique characteristic that enhances biological reactions by their ability to translocate oxygen from the shoots to the roots (Armstrong 1967). This oxygen transport mechanism results in aerobic microzones in the otherwise anaerobic rhizosphere or root zone. The presence of oxygen in the rhizosphere stimulates both the decomposition of organic matter and the growth of nitrifying bacteria. Nitrate formed as a result of nitrification can diffuse or percolate to the anoxic/anaerobic zones where it will be removed from the system by denitrification (Gersberg et al. 1986).

In many areas of the southern U.S. and north, natural wetlands have historically been used as convenient receiving water for wastewater discharges. In 14 states inventoried by the EPA, 326 discharges to surface waters that could be classified as wetlands have been documented (WPCF 1990). Several natural wetlands have been extensively studied, and the ability of wetlands to reduce BOD, nutrients, and metals also has been well documented (Kadlec and Kadlec 1979, Nixon et al. 1986). Tables II-1 (Reed et al. 1979) and II-2 (Hyde et al. 1984) show some typical performance data from natural wetlands (EPA 1988). These data show a very broad range of treatment efficiencies, but this is expected because of the diversity of wetland systems. These data show the potential of natural wetlands for wastewater treatment and water quality improvement.

iii. Natural Wetlands And Legislation

Even though, as discussed above, wetlands have a great potential assimilative capacity for BOD, TSS, and nutrients, they are considered to be surface waters of the United States and can only legally be used for advanced wastewater treatment. Wetlands fall under the jurisdiction of the Clean Water Act, which has a primary objective to assure that designated in stream uses and natural processes are maintained and protected. Wetlands must be considered as receiving systems, and not as treatment systems and a minimum of secondary treatment must be provided (Richardson et al. 1987, Dodd et al. 1986).

The EPA, for the most part, supports the use of natural wetlands for wastewater treatment since they appear to be very cost effective and potentially may have environmentally positive aspects (e.g., wetland restoration). The EPA has even provided funding for research into the wetland treatment systems and is currently examining the development of water quality criteria for wetlands which would receive wastewater discharges (Davis et al. 1987). Only two states, Florida and Wisconsin, have developed water quality criteria for wetlands. Water quality standards are necessary to provide the crucial guidance that designers must have, how much change in water quality will be allowed as result of a wastewater discharge. As a result of the huge diversity of wetlands, and even the diversity within a wetland, water quality criteria will be very difficult to develop (Davis et al. 1987).

Lack of water quality criteria is not the only problem with using wetlands for wastewater treatment. Wetlands are protected areas and any physical alteration has to be studied extensively for potential impacts to the hydrology and plants and animals. If any alterations are required such as channelization or building of dikes to facilitate discharge, permitting must be obtained from the U.S. Army Corps of Engineers under section 404 of the CWA. A modification to an existing wetland also requires review, for

example, under the Endangered Species Act, The National Environmental Policy Act, and any local laws and requirements dealing with wetlands (Davis et al. 1987)

Not only are there tremendous regulatory obstacles to be overcome in using wetlands for wastewater, there are no consistent design guidelines because of the diversity of wetlands. Table II-3 (Richardson and Davis 1987) is a summary of published guidelines for hydraulic loadings of wastewater into wetlands. As can be seen, there is a tremendous range of loadings. It is apparent from the data, because of the diversity of wetland types, that use of "generic" loading limits is inappropriate (Richardson and Davis 1987).

In summary, even though natural wetlands have a tremendous potential for assimilating wastewater, their use for water quality improvement is currently not practical. The huge amount of protective legislation and the lack of existing water quality criteria and design guidance make their widespread use unfeasible at the present time.

B. SFCW Background And Process Description

i. Introduction

The regulatory problems associated with natural wetlands can be avoided by constructing a wetland where one did not exist before. Constructing a wetland treatment system allows a designer to optimize wastewater treatment by enhancing the natural treatment capacity (e.g., high denitrification rates) of these systems, while minimizing the potential negative aspects such as large acreage requirements and low phosphorus removal potential (Reed et al. 1988). Constructing an artificial wetland also offers the advantage of site selection, flexibility in sizing, fewer user conflicts with conservation goals, and most importantly, control over the hydraulic pathways and retention time (Richardson and Davis 1987).

Of the five types of natural wetlands discussed earlier; saltwater swamps, freshwater swamps, coastal salt marshes, freshwater marshes, and bogs; constructed wetlands that simulate marshes with herbaceous emergent and submergent plants have

the most promise for wastewater treatment. Swamps may require 5 - 20 years for the full development and growth of their water tolerant woody plants before significant operational performance can be achieved. Bogs are difficult to establish, have limited retention capacity , are highly intolerant of fluctuating water levels, and are likely to become marshes if nutrient inputs are increased. On the other hand, typical freshwater marshes and associated vegetation are adapted to fluctuating water and nutrient levels and are more tolerant of high pollutant concentrations (Hammer and Bastian 1989).

ii. Types of Artificial Wetlands

There are two main types of constructed or artificial wetlands for wastewater treatment, and they are categorized by their flow regime through the wetland bed. Free water surface (FWS) constructed wetlands (Figure II-1) have water depths of 0.33 to 2 ft (0.1 to 0.6 m), and flow in a shallow bed or channel with relatively impermeable bottom soil or subsurface barrier. Wastewater is typically applied continuously, and treatment occurs as the water flows slowly through the stems and roots of the emergent vegetation. SFCW (Figure II-2 (EPA 1988)) contain a permeable media (soil, sand, or gravel) which supports the same types of emergent vegetation as the FWS systems, but the water level in the bed is maintained below the top of the media. Typical bed depths vary from 12 - 30 in (0.3 - 0.76m) depending on plant species chosen and potential root penetration. Different emergent hydrophytes have varying root penetration potential. Subsurface systems have also been called rock reed filters (RRF), microbial rock reed filters (MRRF), vegetative submerged beds (VSB), and wastewater treatment by the root zone method (RZM) (Watson 1992, Reed and Brown 1992).

SFCW have several advantages over FWS systems. Since the wastewater is maintained below the media surface, there is little risk of odors, public exposure, or insect vectors. Also in SFCW, the media provides greater available surface area for microbial activity and treatment than in FWS wetlands, resulting in higher potential reaction rates. This results in smaller subsurface flow systems than FWS systems

designed for the same loading (Reed et al. 1992). Approximately 150 operating constructed wetlands system exist in the U.S., and of these, 98 are subsurface flow systems (Reed and Brown 1992). Most of the past and current work taking place in Europe and the U.S. is being directed towards subsurface flow systems. As a result of the advantages of subsurface flow systems over FWS systems and the availability of information, this report will discuss SFCW systems.

iii. Artificial Wetlands Constraints

Though artificial wetlands have potential as wastewater treatment systems, they also have some constraints that need to be considered (EPA 1988):

- Geographical limitations of plant species, as well as the potential that a newly introduced plant species will become a nuisance or an agricultural competitor.
- Constructed wetlands that discharge to surface water require 4 to 10 times more land area than a conventional wastewater treatment facility. Zero discharge constructed wetlands require 10 to 100 times the area of conventional wastewater treatment systems.
- Plant biomass harvesting is constrained by high plant moisture content and wetland configuration.
- Some types of constructed wetlands may provide breeding grounds for disease producing organisms and insects and may generate odors if not properly managed.

iv. Background/History

The study of the use of artificial wetlands and aquatic plants for wastewater treatment has been going on in Europe since the 1950's. The earliest work started with Dr. Kathe Seidel at the Max Planck Institute in Krefeld Germany. The process, known as the Max Planck Institute Process (MPIP), features several stages following primary settling. The system utilizes either vertical or long and narrow horizontal flow beds, with a top layer of sand with wetland vegetation followed by bottom layers of gravel. The initial stages remove colloidal matter and carbonaceous BOD with the subsequent stages

used for nitrification and polishing. Design is based on loading factors and guidelines developed from initial experiments and operational experience (Watson 1992).

The early work of Seidel was built upon by Professor Reinhold Kickuth at the University of Hesson in Germany who called his approach the root zone method (RZM). The RZM is characterized by reeds planted in selected (light clay or heavy soil) or in-situ soils with calcium, iron, or aluminum additives to improve soil structure. The minerals were added to strengthen pores which develop, in theory, from dead decaying roots, and provide a cation source for coprecipitation of phosphates. The flow through the bed is horizontal and is distributed and collected from stone trenches containing perforated pipe on both sides of the bed. Hydraulic design is based on Darcy's law and surface area is based on first order reaction kinetics (Watson 1992). Kickuth claimed that his RZM could attain 90% removal of BOD, TSS, N, and P. He also claimed, after a maturing period of 3-5 years, that hydraulic conductivity of 10^{-3} m/s would be attainable through the soil media. The increased conductivity would result from root penetration and a pore structure developed from dead and decaying roots. Overall, performance claims for the system have not been attained (Brix 1987, Findlater et al. 1990, Conley et al. 1991). Failure has primarily resulted from Kickuth's claim of a predicted increase of soil permeabilities with time, resulting in surface flow over most of the wetland area. Also, Kickuth's estimates on the amount of oxygen transported to the rhizosphere is considered to be to optimistic(Brix and Schierup 1989, Schierup and Brix 1990).

Hundreds of RZM type treatment systems were installed throughout Europe based upon Kickuth's initial claims, with most of the systems experiencing surface flow. As seen in Appendix A, BOD and TSS removals were acceptable, but less than expectations. Nitrogen removal was far short of expectations. Danish authorities were so disappointed with the RZM that they officially disassociated themselves from the method (Brix and Schierup 1989).

As a result of the problems with the original RZM design criteria, the European Community/Water Pollution Control Association (EC/EWPCA Cooper 1990) published a set of new design criteria to follow to ensure that this promising technology would not fall into complete disregard. The design guidelines put forth by the European Community are based upon the ideas of Kickuth but modified to account for problems in systems that were built according to his theories. Kickuth recommended a BOD_5 design load of 180 kg/ha-d (162 lb/ac-d). In practice this number was too optimistic and some systems that loaded at this level had problems with BOD_5 (Brix 1987, Brix and Schierup 1989). The published guidelines of 80 kg/ha-d were much more conservative. The original RZM recommended using existing site soils for bed media, with design hydraulic conductivities as high as 10^{-3} to 10^{-4} m/s, based on Kickuth's theory that hydraulic conductivity would increase as the bed matures and roots and rhizomes develop (Cooper and Boon 1987). This claim has been disputed on the basis of calculations made using Kickuth's assumptions and on the data available from existing systems (Bucksteeg 1985). In practice, the systems based on the assumptions of increased hydraulic conductivity experienced massive surface flow. This experience led to the recommendations of using the in-situ hydraulic conductivity in design or to use gravel. The original RZM also recommended bottom slopes of 2% to 8%. The slopes proved to be excessive and resulted in short circuiting and the inability to flood the wetland. Periodic flooding is recommended to control weeds. The European design standards also recommend that these systems not be installed for nutrient removal. It was originally claimed that nutrient removals of 90% could be achieved by the RZM, but actual removals were in the 20% - 50% range.

The poor performance of the RZM can be attributed to surface run-off (low permeability of the soil), which prevents the sewage from getting into the rhizosphere, and insufficient release of oxygen from the root system of the reeds to assure significant nitrification (Brix and Schierup 1989). The use of the more conservative design

approach put forth in the European design manual and more study of the hydraulics and oxygen transfer may lead to systems that can live up to earlier claims.

In the U.S., the use of artificial wetlands for wastewater treatment was introduced and accepted as a viable treatment option at the first international conference on biological control of water pollution held at the University of Pennsylvania in 1976. At this conference the work of Seidel (1976) and her work at the Max Planck Institute was presented along with the work of Wolverton and Mcdonald (1976) of the National Aeronautics and Space Administration (NASA). NASA has been one of the leaders in developing this technology because of its use in future Closed Ecological Life Support Systems (CELSS) (Wolverton 1980, 1987). Most of the early work in the U.S. was accomplished by B. C. Wolverton and R. M. Gersberg.

Wolverton's work with emergent plants began in the early 1980's with experimental bench scale trays batch loaded with wastewater and drained after 12 to 48 hours. The trays were filled with rock or gravel and emergent aquatic plants and achieved excellent removal of BOD, TSS, and NH_3 and moderate removal of phosphorus (Reed et al. 1992, Wolverton 1983). Wolverton's approach was a combination of his early work with hydroponics (the growing of plants in a nutrient solution and without soil) and microbial rock filters. The combination of hydroponics and microbial rock filters result in a symbiotic relationship between plants, plant roots, the microorganisms, and the adjacent rocks (Wolverton 1988). Wolverton's microbial rock plant filter (MRPF) is a highly managed system that requires the placement of plants in a highly structured pattern to achieve a desired bed root volume. The system is harvested and operated according to a scheduled program designed to maintain hydraulic subsurface plug flow by controlling root growth in the bed, while providing a sufficient volume of wastewater to sustain hydroponic growth . These systems are achieving consistently low BOD, TSS, and NH_3-N effluents of 5/5/5 and 10/15/5 at Benton and Sibley, LA (Jones 1992).

Gersberg's work in Santee, CA was conducted over a period of several years (early to mid 80's) in a large scale, continuous flow, field experiment. His work was an assessment of the ability of the three most common artificial wetland plants, Scirpus validus (bulrush), Phragmites communis (common reed), and Typha latifola (cattail), to remove nitrogen, BOD, and TSS from primary municipal wastewaters. He was able to achieve ammonia removal efficiencies of 94% (bulrushes), 78% (reeds), and 28% (cattails) and BOD removal efficiencies of 96%, 81%, and 74%, respectively. The high removal efficiencies were directly attributable to the root penetration potential of the different plants. Of the three plant species, cattail had the shallowest root zone with most of the roots confined to the top 30 cm of substrate. The root zone of the bulrushes and the reeds extended to >60 cm and 76 cm, respectively. Therefore, oxidized conditions favoring BOD removal and nitrification were probably more favorable in these beds (Gersberg et al. 1986).

As in Europe, the initial studies of SFCW proved highly favorable and also as in Europe the technology was grasped with enthusiasm, but many systems were installed that did not live up to expectations. The problems, as in Europe, resulted from poor hydraulic performance and insufficient oxygen transfer to the media. The hydraulic performance of the SFCW systems is a result of poor hydraulic design and/or poor construction practices. At four sites in Louisiana that were experiencing surface flow, the cause was traced to inorganics clogging the media introduced during construction. Hydraulic design should improve with more experience and the introduction of adjustable outlet structures. Oxygen transfer can be improved by matching bed depth to plant species to ensure the the wastewater comes in contact with the full root zone (Reed et al. 1992).

v. Summary of Current Design Approaches

Both the EPA (1988) and the Water Pollution Control Federation (1990) published manuals containing general design guidance for constructed wetlands. Both

manuals depend heavily on case histories and performance data from past work and existing operating systems. As seen in Table II-4 (Reed et al.1992), depending on whose past work is used for a reference, designers are faced with a huge range of possible design criteria, and it becomes apparent that there is no consensus on how these systems should be designed (Reed et al. 1992).

In recognition of both the potential of constructed wetlands and the lack of design consensus, various offices within the EPA are sponsoring research efforts to better understand the capabilities and limitations of the use of constructed wetlands for wastewater treatment (Reed et al. 1992). A summary of current design and performance expectations are presented as follows:

BOD

BOD is removed by both physical and biological mechanisms. Physically BOD is removed by particulate settling and entrapment in the void spaces of the gravel or rock media, and soluble BOD is removed by microbial growth on the media surfaces and plant rhizomes as they penetrate the bed. BOD removal is believed to proceed very rapidly with most removal occurring within the first day and little after 7.5 days. Most of the bed is believed to be anaerobic with aerobic microzones existing adjacent to the surface of the plant roots.

A first order plug flow model is believed to apply to these systems and is used by most engineers in the design of these systems:

$$Ce/Co = \exp(-K_T t)$$

Where: Ce = Effluent BOD_5 (mg/L)
Co = Influent BOD_5 (mg/L)
K_T = temperature dependent rate constant (day^{-1})
$K_T = K_{20}Ø^{(T-20)}$
$Ø = 1.06$
t = hydraulic residence time (day)

K_{20} is given from several sources and is recommended to be 1.104 days $^{-1}$.

The surface area of the bed is given by:

$$A_S = Q\,[\ln(Co/Ce)] \,/\, (K_T d n)$$

where:

A_S = bed surface area, m^2 (ft^2)
Q = average flow through bed, m^3/d (ft^3/d)
d = bed depth, m (ft)
n = effective porosity of media.

The bed depth should not exceed the potential root penetration of the vegetation to be used to ensure sufficient oxygen transfer to the media (Reed et al. 1992).

TSS

Suspended solids removal is very effective in SFCW with most of the removal occurring within the first few meters of travel from the inlet zone. As with BOD removal, most TSS removal occurs within the first day of retention. Generally, if a bed is sufficiently designed for BOD removal and does not experience massive surface flow or short circuiting, TSS removals will be extremely high (Reed et al. 1992)

Nitrogen

SFCW have the potential to be highly effective in removing nitrogen. Nitrogen is removed primarily by microorganisms in the rhizosphere of the vegetation. Plant uptake is believed to account for very little (< 10%) of the removal of nitrogen in SF constructed wetlands (Reed et al. 1992). Nitrogen removal and design approaches for nitrogen removal are discussed in detail in Section III of this report.

Phosphorus

Phosphorus in natural treatment systems is removed by adsorption and precipitation reactions with clay, aluminum, iron, and calcium compounds (EPA 1981). Unless a soil is used, as opposed to rock or gravel, that is high in these minerals, SF constructed wetlands are ineffective at phosphorus removal. Because of low hydraulic conductivity, soils are not recommended for constructed wetlands and if significant phosphorus removal is a project requirement, then very large land areas or alternative treatment will be required (Reed et al. 1992).

Fecal Coliform

Generally SFCW systems are only capable of a 10 fold reduction in fecal coliforms which is not enough to satisfy discharge requirements of < 200/100 ml. It is recommended that some sort of final disinfection be utilized before final discharge (Reed et al. 1992).

Hydraulics

As discussed earlier, one one of the major problems with SFCW is surface flow resulting from either poor hydraulic design or construction practices that allow the introduction of fines that can potentially clog up the pore structure of the rock or gravel media. It is common practice to design SFCW using Darcy's law as follows:

$$Q = k_s AS$$

where:

Q = flow per unit time, m^3/d (ft^3/d)
ks = hydraulic conductivity of a unit area of medium perpendicular to flow, m^3/m^2-d
A = cross sectional area, m^2 (ft^2)
S = hydraulic gradient dh/dl, m/m (ft/ft).

Q should be the average flow in the system to account for any gains or losses due to precipitation, evaporation or seepage.

Most of the problems in current operational systems are believed to result from inadequate hydraulic gradient caused by system design and configuration. It is recommended that the beds be constructed with a sloping bottom and/or adjustable outlet works that would allow the water level to be controlled within the bed. The aspect ratio (L:W) needs to be kept relatively low to ensure sufficient hydraulic gradient, since the maximum available hydraulic gradient is equal to the depth of the bed divided by its length, and 0.4:1 to 3:1 is recommended. Finally, it is recommended that only 50% of the apparent hydraulic conductivity of the bed media be used as a factor of safety to

ensure subsurface flow (EPA 1981). This number is consistent with that used in land treatment systems (Reed et al. 1992).

Table II-1. Percent removal for several pollutants from secondary effluent in natural wetlands.

Pollutant	Percent Removal
BOD_5	70-96
Suspended Solids	60-90
Nitrogen	40-90
Phosphorus	Seasonal

(After Reed et al. 1979)

Table II-2. Summary of nutrient removal from natural wetlands.

Location	Flow, m^3/d	Type	TDP[a]	NH_3	NO_3-N	TN[b]
Brillion, WI	757	Marsh	13	-	51	-
Houghton Lake, MI	379	Peatland	95	71	99	-
Wildwood, FL	946	Swamp	98	-	-	90
Concord, MA	2,309	Marsh	47	58	20	-
Bellaire, MI	1,136	Peatland	88	-	-	84
Dundas, Ontario, Canada	-	Marsh	80	-	-	65
Home Park, Fl	227	Cypress	91	-	-	89

a - Total Dissolved Phosphorus
b - Total Nitrogen

(After Hyde et al. 1984)

Table II-3. Summary of guidelines for hydraulic loading of wastewater into natural wetlands.

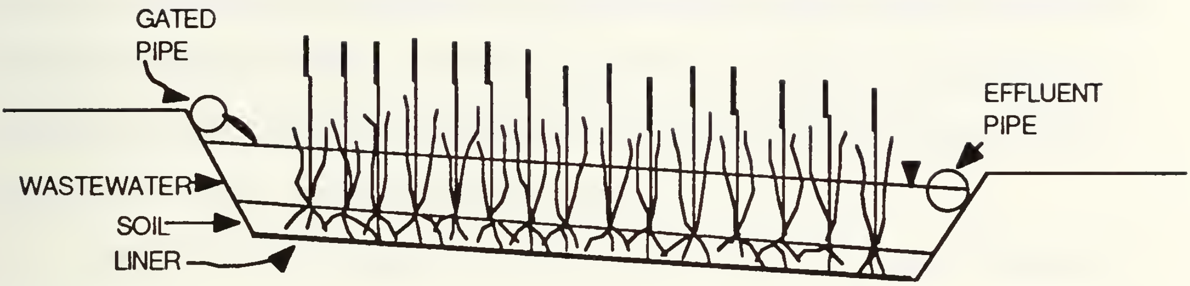
Author	Loading Rate	Rationale
Kadlec and Tilton (1979)	2.5 cm/wk	"2.5 cm or so per week... is not unreasonable from the viewpoint of the natural precipitation input to the typical wetland." Will provide adequate treatment of nutrients and suspended solids.
Richardson and Nichols (1985)	0.76-2.9 cm/wk	For 50% to 75% removal of N & P.
Stowell et al. (1981)	<u>BOD</u> 261 cm/wk <u>N</u> 26.2 cm/wk	Based on BOD removal from secondary effluent (30 mg/L) and 80% removal efficiency. Based on N removal from secondary effluent (30 mg/L).
Tchobanoglous (1980)	10.9-21.8 cm/wk	"N & P removal is uncertain and may require larger areas of significant removal."
EPA (1985)	2.5 cm/wk	

(After Richardson and Davis 1987)

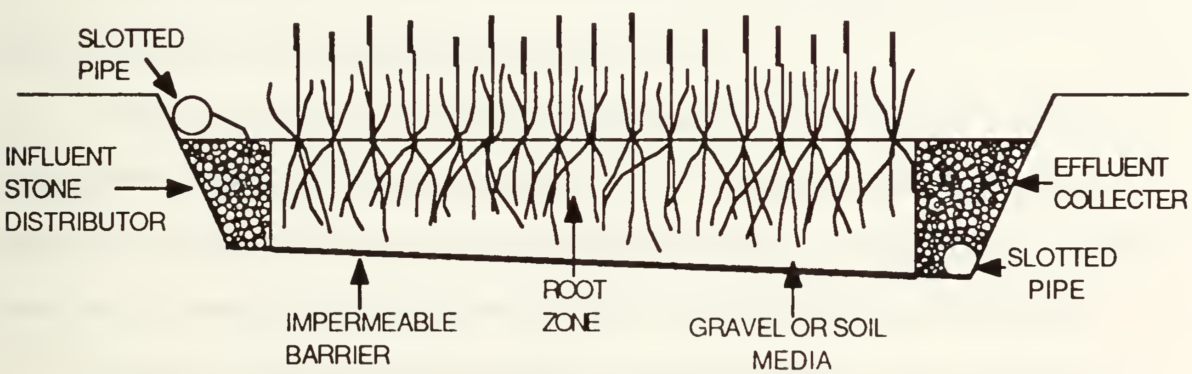
Table II-4. Summary of guidelines for loadings of wastewater into SFCW.

Author	Hydraulic Loading (cm/d)	BOD_5 Loading (kg/ha-d)
Kickuth (Boon 1983)	-	180
EC/EWPCA (1990)	4	80
Wolverton (1983)	8	58
Gersberg et al. (1986)	5	55

(After Reed et al. 1992)



<u>**Figure II-1**</u>. Typical cross section FWS wetland.



<u>**Figure II-2**</u>. Typical cross section SFCW (After EPA 1988).

SECTION III NITRIFICATION/DENITRIFICATION

A. Introduction

As discussed previously, subsurface flow constructed wetlands are highly effective in consistently removing BOD and TSS from municipal wastewaters. Studies also show that constructed wetlands have the potential to remove nitrogen (Gersberg et al. 1983,1984,1986; Rogers et al. 1991; Burgoon et al. 1991), but performance in practice has been inconsistent at best (Brix and Schierup 1989, Chalk and Whealon 1989, Conley 1991, Schierup and Brix 1990).

As can be seen in Figure III-1 (Tchobanoglous and Burton 1991) showing the behavior of nitrogen in natural treatment systems, the nitrogen in wastewater once introduced has potentially several mechanisms (e.g., volatilization, denitrification, biomass uptake, etc.) for removal, resulting in nitrogen loss from the natural system. The removal mechanism depends on the form of nitrogen present. Typically nitrogen in most wastewaters is in the form of organic-N, NH_3 (ammonia), and NO_3^- (nitrate), with nitrate as the predominant form if the wastewater has undergone some form of advanced treatment (Tchobanoglous and Burton 1991).

Organic nitrogen, as part of the suspended solids, is removed by sedimentation and filtration. Solid phase organic nitrogen may be incorporated directly, physically or chemically, into the soil or hydrolyzed to soluble amino acids that may undergo further breakdown to release ionized ammonia (NH_4+) (Tchobanoglous and Burton 1991).

Ammonia nitrogen has several pathways through a natural system. Some of the soluble ammonia can be volatilized directly into the atmosphere as ammonia gas. Most of the influent and converted ammonia in a natural system is adsorbed onto the soil particles where it is made available to plants and microorganisms for nutrient uptake. Ammonia is also converted to nitrate by microorganisms through the aerobic biological process of nitrification (Tchobanoglous and Burton 1991).

Nitrate nitrogen, because it is a negatively charged, is not adsorbed onto soil particles, remains in solution, and may be transported to groundwater. If nitrate is not removed by plant uptake or denitrification, it may cause problems with groundwater supplies. Nitrate uptake by plants only occurs in the area of the root zone and only during the limited growing period. In most natural treatment systems, for plant uptake of nitrogen to be truly effective the vegetation must be harvested or the nitrogen retained in the biomass will be recycled back into the system (Tchobanoglous and Burton 1991).

Facultative bacteria under anoxic conditions also remove nitrate by biological denitrification. This process results in the nitrogen being released to the atmosphere in the form of nitric oxide (NO), nitrous oxide (N_2O) or molecular nitrogen gas (N_2). It is interesting to note, and will be discussed further later, that it is not necessary for the entire system to be anoxic for denitrification to occur. Anoxic and aerobic microzones are believed to exist adjacent to each other in most natural treatment systems in the soil/water/plant interfaces (Tchobanoglous and Burton 1991).

Subsurface flow constructed wetlands (SFCW), because of the way they are constructed and operated, limit some of the nitrogen transformations and removal mechanisms that are present in other natural treatment systems. SFCW typically are built with a liner, so nitrate percolating to the groundwater and being removed from the system does not occur. As a result of the subsurface flow conditions, ammonia volatilization losses are negligible, and because of the typically used gravel media, adsorption is minimal. Organic nitrogen hydrolization, biological sequential nitrification/denitrification, and plant/microorganism nutrient uptake still occur in SFCW with denitrification and plant uptake the major removal mechanisms of nitrogen from wastewaters.

Even though there are some studies that show plant uptake accounting for as much as 90% of the nitrogen removed from wastewaters (Rogers et al. 1991 and Breen

1990), it is the general consensus that biological nitrification/denitrification is the major removal mechanism of nitrogen in SFCW systems with plant uptake accounting for less than 10-15% (Gersberg et al. 1986, Reed et al. 1988, Herskowitz et al. 1987, Gearheart et al. 1985). This section of this report discusses both biological nitrification/denitrification as well as the possibility of significant plant uptake of nitrogen as possible removal mechanisms in SFCW.

B. Nitrification/Denitrification

i. Process Description

Nitrification/denitrification is a sequential two step process in which ammonia is first aerobically oxidized to nitrate by chemoautorophic nitrifying bacteria and then, secondly, the nitrate is reduced to nitrogen gas or nitrous oxide by facultative heterotrophic denitrifying bacteria. The denitrifying bacteria use the nitrate and nitrite in place of free oxygen as the final respiratory electron acceptors to carry out the oxidation of carbonaceous organic substrates (Gersberg et al. 1984).

Nitrifying bacteria are widely distributed in soil and water. Two groups of bacteria, Nitrosomonas and Nitrobacter, working sequentially are responsible for nitrification. Nitrosomonas oxidize ammonia to nitrite and Nitrobacter oxidize nitrite to nitrate. The nitrifying bacteria utilize ammonia and nitrite oxidation as an energy source and obtain their organic carbon via carbon dioxide fixation (converting inorganic carbon to organic) (Brock and Madigan 1991).

Denitrification is accomplished by bacteria (e.g., Achromobacter, Aerobacter, Alcaligenes, Bacillus, and Pseudomouas) that are capable of dissimilatory nitrate reduction. First nitrate is converted to nitrite which is then converted to nitric oxide, nitrous oxide and nitrogen gas. In the presence of oxygen, these bacteria use oxygen for respiration, but, in the absence of oxygen, enzymes are produced that modify the aerobic pathways to utilize nitrate and nitrite as electron acceptors in place of oxygen. These

bacteria use organic carbon for both energy and a carbon source (Tchobanoglous and Burton 1991).

Sequential biological nitrification/denitrification is pH and temperature dependent. The reactions occur faster at pH ≥ 7 and at pH ≤ 5 the reaction is inhibited. Studies show that denitrification rates proceed very slow at low temperatures, 2^{o} C, and increase to a maximum rate at about the 60^{o} -65^{o} C range. For every 10^{o} increase in temperature between 11^{o} C and 35^{o} C, rates have been shown to double (Nichols 1983, Stanford 1975).

ii. Rhizome Water/Sediment Interface

The key to driving the sequential nitrification/denitrification process is supplying oxygen to the rhizome (root zone) so that nitrification can occur. Given that carbon is not limiting for denitrification, the process proceeds rapidly and is limited by the supply of nitrate, with the denitrification rate independent of nitrate concentration over a wide range of conditions (Nichols 1983, Bowmer 1987).

As discussed earlier, wetland plants have the unique ability to translocate oxygen to their roots with some oxygen diffusing into the surrounding sediment or water. Oxygen movement within wetland plants occurs because of and is dependent upon the presence of tissue called aerenchyma. This tissue is unique in that it has a low resistence to gaseous diffusion (Good and Patrick 1989). Figure III-2 (Good and Patrick 1989), is a typical cross section of a plant root that shows the aerenchyma.

This ability to transport oxygen is very important for plants living in the reduced (anoxic) environment present in wetlands. Oxidation in the root zone removes soil toxins such as H_2S and reduced Fe and Mn (Armstrong 1972, Gambrell and Patrick 1978, Ponnamperuma 1965), and it is also necessary for aerobic respiration in the plant's root cells. Aerobic respiration is much more efficient in producing energy than anaerobic processes such as lactic acid or alcohol fermentation (Good and Patrick 1989) . Evidence of internal oxygen transport and subsequent diffusion into the surrounding

sediment is seen in the observed oxidation of iron on wetland plants root systems. The reduced form of iron is soluble in water while the oxidized form is not, and therefore it precipitates onto the root surface (Gambrell and Patrick 1978, Bach and Hossner 1977, Good and Patrick 1989).

The oxygen diffusing from the plant roots is believed to be utilized by nitrifying bacteria. As can be seen in Figure III-2, ammonia moves into the aerobic zone via concentration gradients and that which is not taken up by the roots is nitrified. The nitrate that is not taken up then is transported by concentration gradient to the anoxic zone where denitrification takes place (Good and Patrick 1989).

C. Artificial Wetlands in Santee CA

One of the most important studies to date in the use of constructed wetlands for treating primary and secondary municipal waste took place at the San Diego Region Water Reclamation Agency in Santee, CA in the early to middle 1980's (Gersberg et al. 1983, 1984, 1986). This work showed the full potential of constructed wetlands for removing BOD, TSS, and nitrogen. Gersberg et al. were able to create the "ideal" constructed wetland, and they obtained outstanding results.

Their wetlands were ideal in that they hydraulically were able to maintain subsurface flow, their vegetation's root zones grew to their full potential, and the oxygen transport capabilities of the plants were maximized. Their artificial wetlands were constructed using gravel as the media. By carefully constructing their beds at a 1% bottom slope and the use of valves and flow meters, they were able to precisely control the flow through the bed to maintain subsurface flow. They were able to achieve root penetrations of 30 cm for cattails (Typha latifola), 60 cm for bulrushes (Scirpus validus), and 76 cm for common reed (Phragmites communis) (Gersberg et al. 1986). These root penetrations are believed to be the maximum attainable (Reed et al. 1992). Also, as a result of the warm and sunny climate typical of the San Diego area, oxygen transport was maximized by convective flow through aeration via plant tissue. It is

believed that oxygen transport is driven by light and temperature and humidity differences between the outside air and the internal plant tissues (Brix 1988, Brix and Schierup 1990, Armstrong and Armstrong 1990, Armstrong et al. 1990). In essence, the aquatic plants are believed to act as natural pumps delivering oxygen, in this case, to the rhizome. These ideal conditions created high removal efficiencies for not only BOD and TSS, but nitrogen as well.

The first set of experiments conducted at Santee CA. (Gersberg et al. 1983), consisted of 14 pilot scale (18.5m x 3.5m x 0.76m) artificial marshes planted with reeds (2 beds), bulrushes (8 beds), cattails (2 beds), and two control beds (no plants) testing secondary effluent applied at various application rates. The beds were first tested without supplemental carbon, and total nitrogen removal was less than 25%. The next phase of the experiment added methanol as a carbon source and 95% removal was achieved at a hydraulic loading rate (HLR) of 16.8 cm/day. In the third phase of the experiment, mulched plant biomass was applied to the surface of the bed, and 60% removal efficiency was achieved at applications of 16.8 cm/day while 86% was achieved at 8.4 cm/day.

The second set of experiments were similar to the first in that secondary effluent was applied with methanol and mulched biomass as a carbon source. In addition, blended primary effluent was added as well, as a carbon source. For this experiment, four large demonstration scale artificial marshes were constructed. Two beds were 71m x 11.6m x 0.76m and two beds were 65.7m x 11.3m x 0.61m. One of each bed size was planted with cattails or bulrushes. These experiments showed similar results with removals of 94% of total nitrogen for HLR of 20 - 25 cm/day for wetlands with methanol added and 89% removals with mulch added at HLR of 8-12 cm/day. Utilizing blended primary effluent as the carbon source with a total HLR of 18 cm/day, they were able to achieve a 77% reduction in total nitrogen with effluents of 5 mg/L, well within secondary requirements.

The third set of experiments looked at the application of primary effluent applied at a HLR of 4.7 cm/ day to four 18.5m x 3.5m x 0.76m artificial wetlands (hydraulic retention time (HRT) 6 days) with one bed each planted with bulrush, common reeds, cattails, and one control bed. Their results showed that ammonia removal efficiencies of 94% (bulrushes), 78% (reeds), and 28% (cattails) as compared to 11% for the control plot. Effluent nitrate levels where 0.99 mg/L, 0.05 mg/L, 0.15mg/L, and 0.09 mg/L, respectively, indicating that most of the ammonia nitrified was removed by denitrification. They calculated that based upon maximum plant nutrient uptake, biomass production could have only accounted for 12%-16%, demonstrating the tremendous sequential nitrification/denitrification capability of constructed wetlands. The differences between the plant species was primarily attributed to the different bed penetrations of the plants' roots resulting in portions of the wastewater not coming in contact with the rhizome and not being sufficiently oxygenated. Some of the differences also may have resulted from different oxygenation capacity of the plants involved.

D. Discussion

Gersberg's work in California represents optimum performance under ideal conditions, but, as discussed earlier, field applications have failed to live up to expectations. Poor nutrient removal can be traced to poor hydraulic performance (either short circuiting or surface flow) and insufficient root penetration of the emergent wetland plants resulting in insufficient oxygenation.

In Europe, poor hydraulic performance resulted from overestimating the hydraulic conductivity of the bed media, and in the U.S. it appears that the problems may stem from poor construction practices and poor hydraulic design. These problems can easily be overcome by tightening construction specifications and incorporating a sufficiently large factor of safety into the expected hydraulic conductivity of the media. For example, Reed et al.(1992) recommend a factor of safety of 2 to ensure subsurface flow whether soil, sand, or gravel is used for the bed media.

Poor root penetration appears to be a bigger problem in the U.S. than in Europe. European operations lower the water level in their beds in the Fall of every year to promote root growth through the full bed depth. This periodic drawdown has not been practiced in the U.S. (Reed et al. 1992). Most of the beds operating in the U.S. are constructed with a bed depth of about 0.6m, but in most cases root penetration is only about half the bed depth resulting in approximately half the flow not coming in contact with the root zone. Reed et al. (1992) conducted a study of 10 SFCW and found only two sites achieving high ammonia removal efficiencies. One was the pilot scale site in Santee, CA and the other was a site in Bear Creek, AL. The Bear Creek site was constructed with a 0.3m bed and the vegetation (cattails) achieved full root penetration. This supports the requirement that to achieve nitrification the wastewater has to come in full contact with the rhizome (Reed et al. 1992).

The consensus is that emergent wetland plants have the ability to transport oxygen from the shoots to the roots; the question remains as to how much oxygen is transported and more importantly how much oxygen is available for the microorganisms to utilize. For example, for reeds (Phragmites), Brix and Schierup (1990) concluded that very little oxygen is available for utilization and reported results of 0.02 g-O_2/m^2-day . At same time, Armstrong et al. (1990) showed results of 4.6 to 12 g-O_2/m^2-day of oxygen available, from reeds, on the average and showed as much as 20 g-O_2/m^2-day could be available under the right conditions. Reed et al. (1992), using Gersberg's et al. (1986) results, estimated that 4.8 g-O_2/m^2-day was available in the Santee, CA reed bed wetlands system. Reed et al. (1992) also reported 45 g O_2/m^2-day by Lawson (1985). This broad range of numbers indicates that there is no consensus among researchers because, as discussed earlier, the amount of oxygen available is very site specific, resulting from the internal plant transport mechanisms being driven by light, temperature and humidity.

E. Nutrient Plant Uptake

The general conclusion concerning nitrogen uptake by wetland vegetation is that it only accounts for 10% - 15% of the nitrogen load applied (Gersberg et al. 1986; EPA 1988; WPCF 1990; Reed et al. 1988,1992; Herskowitz et al. 1987; Brix 1987).

There are some studies in Australia (Rogers et al. 1991, Breen 1990, Breen and Chick 1989) that dispute the now general assumption that plant uptake of nitrogen is negligible. These studies consisted of bench scale constructed wetlands (25 liter plastic buckets) utilizing a vertical upflow or downflow hydraulic format as opposed to horizontal flow through format. Rogers et al. (1991) were able to achieve (utilizing bulrush) 95% total nitrogen removal, of which they attributed 80% to plant uptake. The loadings and removal efficiencies obtained were comparable to that of Gersberg et al. (1986) at 1.3 g-N/m^2-day and 94% removal efficiency. They were later able to achieve 91% removal of total nitrogen with a loading of 7.8 g-N/m^2-day (HRT of 1.75 days), of which they attributed 90% to plant uptake.

A similar bench scale study (Burgoon et al. 1991) utilized bulrush with batch loads of 0.6 and 2.2 g-N/m^2-day. They were able to achieve removals of 88% and 79% (HRT of 2 days), respectively. Their mass balance showed that 30% of the lighter load went to biomass while only 8% of the heavier load did. They concluded that as loading increased the amount of nitrogen in the plant biomass increased, but the percent of total nitrogen removed decreased and that plants competed effectively with nitrifying bacteria for available nitrogen.

Both studies showed that bulrush will absorb and store nitrogen, but Burgoon et al.'s (1991) results are more reasonable and much more consistent with Gersberg et al. (1986) and past work with constructed wetlands. Rogers' et al. (1991) nitrogen removal of 91% at a HRT of 1.75 days loaded with 7.8 g-N/m^2-day, which is six times a typical primary wastewater loading of nitrogen, is inconsistent with all other performance data reviewed for this report. It is also questionable whether bench scale

studies with vertical or batch loadings are comparable to a full scale study utilizing a horizontal flow pattern.

F. SFCW Design For Nitrification/Denitrification

Two design approaches will be presented, one developed by Reed (1992) and a more conservative design approach developed by the WPCF (1990). Reed's approach is based on estimates of the available oxygen for nitrification in a constructed wetland bed with a fully developed root zone. From the assumption that it takes 5 mg of oxygen to nitrify 1 mg of ammonia, the required bed area is calculated from the assumed oxygenation capacity of the selected vegetation.

The WPCF's approach is based upon a regression analysis of data collected on the performance of both natural and constructed wetlands for the removal of ammonia or total nitrogen. The WPCF approach uses these equations as follows:

$$(NH_3) \quad A_s = (100Q)/\exp[1.527\ln Ce - 1.050\ln Co + 1.69)]$$

$$(TN) \quad A_s = (100Q)/(0.645Ce - .125Co + 1.129)$$

where:
As = bed surface area, m^2
Q = average flow through bed, m^3/d
Co = influent concentration, mg/L
Ce = effluent concentration, mg/L.

The WPCF recommends that a designer use a minimum size area of 4 ha/1000 m^3-day and a aspect ratio (L:W) of 2:1 or greater. The recommended minimum area of 4 ha/1000 m^3-day is based on performance data and appears to be a minimum requirement to achieve significant nitrogen reduction.

The following is an example utilizing design criteria for the Bear Creek, AL constructed wetland. As discussed earlier, the Bear Creek wetland is one of the very few SFCW in the U.S. that achieved maximum root penetration of the bed and significant ammonia reduction. This wetland was constructed at Phillips High School to polish the effluent from the school's extended aeration package treatment plant. The Bear Creek wetland is 22,000 ft^2 (2041 m^2) and designed for a flow of 20,000 gal/day (75.7

m^3/d). The average influent ammonia concentration is 11 mg/L with a high of 20 mg/L. The NPDES permit requirement for ammonia is 8 mg/L. For this example, the high figure of 20 mg/L will be used for the influent concentration and 8 mg/L for the effluent requirement. The bed depth of this wetland is 12 in and the vegetation is primarily cattails.

Utilizing the WPCF approach:

$$A_s = (100)(75.7)/\exp[(1.527)(\ln 8)-(1.050)(\ln 20)+1.69]=1356 m^2 \ (0.14 \ ha)$$

The minimum area requirement recommended by the WPCF is 4 ha/1000m^3-day. Therefore the required area is:

$$A_s = (4 \ ha/1000m^3\text{-day})(75.7 \ m^3/d) = 3028 \ m^2 \ (0.3 \ ha).$$

. Utilizing the Reed approach and assuming that the cattail bed delivers 2.1 g O_2/m^2-day (Reed et al. 1992):

$$\text{Oxygen required} = (20 - 8)mg/L \ (75.7 \ m^3/day)(5mg(O_2)/mg(NH_3))$$

$$= 4560 \ g\text{-}O_2/day$$

$$\text{Nitrification Area required} = 4560/2.1 = 2171 \ m^2$$

The Reed equation does a very good job,in this example, of predicting the required size of a SFCW when full bed penetration is achieved. The actual Bear Creek wetland is 2041 m^2, has achieved full bed root penetration, and does perform within the parameters of this example.

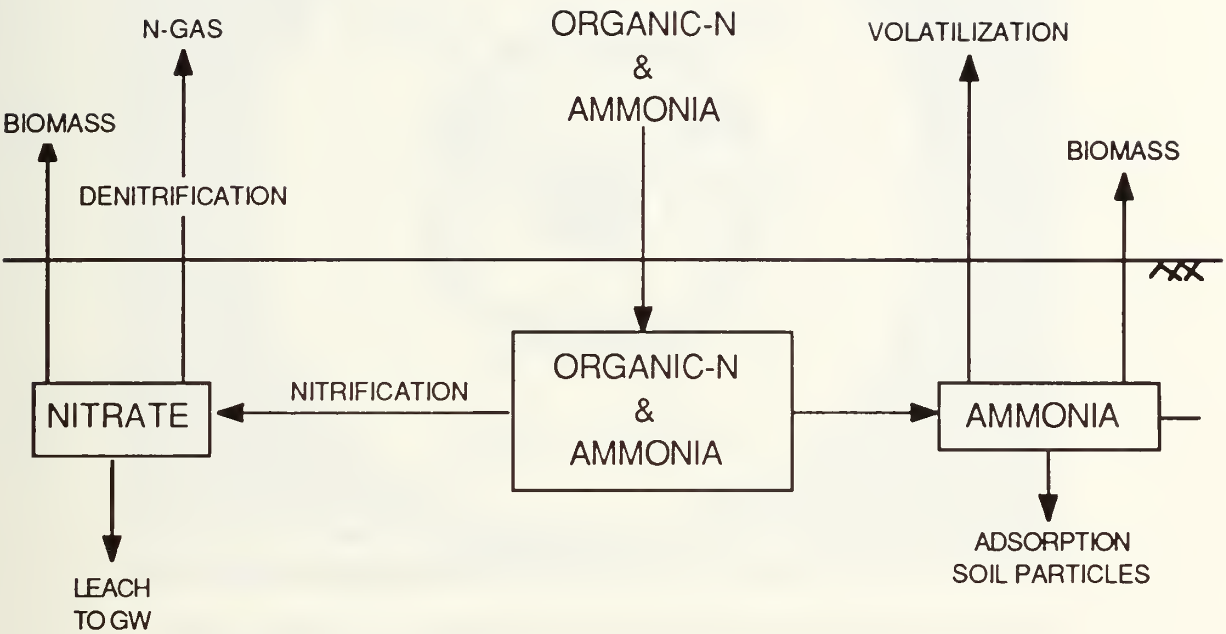


Figure III-1. Nitrogen behavior in natural treatment systems. (After Tchobanouglous and Burton 1991)

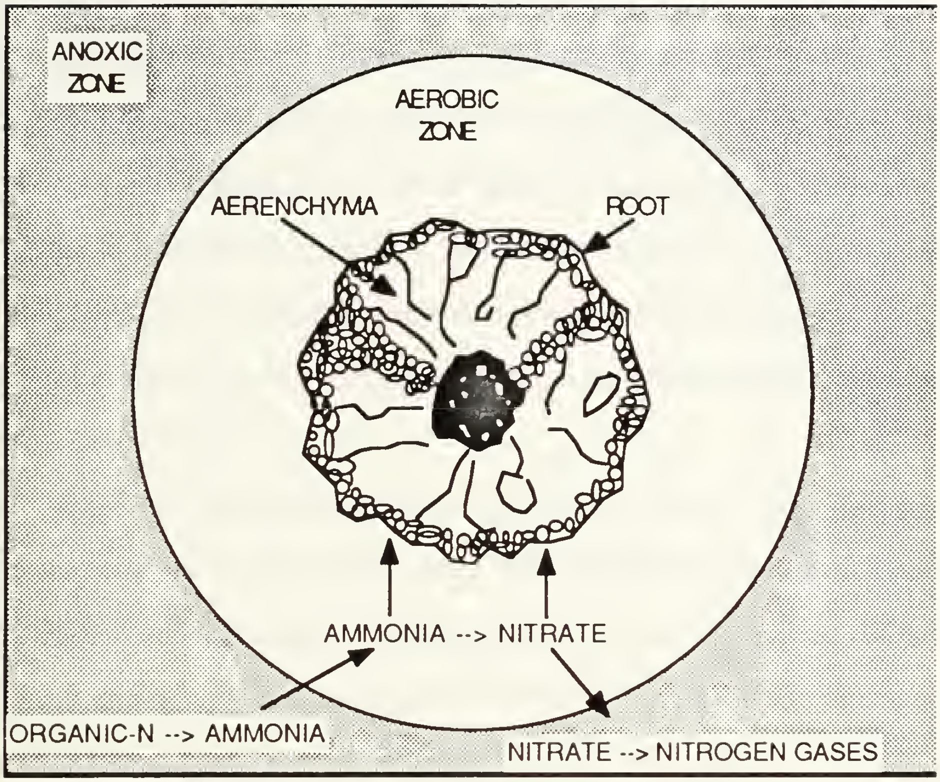


Figure III-2. Cross section of a wetland root in an anoxic sediment. (Good and Patrick 1989)

SECTION IV OXYGEN TRANSPORT

A. Introduction

As previously discussed, the limiting step in the sequential nitrification/denitrification process in SFCW is nitrification, which is itself limited by the amount of oxygen available in the root zones of the aquatic plants. Wetland plants have the ability to transport oxygen from their leaves to their root structures via internal tissues that have a relatively low resistance to gas phase diffusion or flow. The development of these tissues is a response to living in saturated wetland soils that are essentially devoid of oxygen.

Soil aeration is primarily controlled by diffusion, and in unsaturated soils the high gas phase oxygen diffusivity ($D_{O_2/air}$ = 0.0205 cm^2/s) promotes soil aeration. In saturated soils and flooded soils the oxygen diffusivity is greatly reduced ($D_{O_2/wet\ soil}$ = 2.267×10^{-5} cm^2/s and $D_{O_2/water}$ = 1×10^{-5} cm^2/s) and anoxic conditions prevail. Flooded wetland soils also can be considered negatively aerated because of the large amounts of oxidizable compounds present (Armstrong 1978).

Wetland plants in a flooded anoxic environment encounter two major problems: they cannot survive without aerobic respiration in their root zone; and the anoxic environment promotes the development of reduced phytotoxins. In response to the problems encountered, wetland plant species developed a unique internal ventilating system that allows the transport of oxygen to the roots to provide for aerobic respiration and the oxidation of toxins adjacent to their root structures. Oxygen stress triggers the development of gas permeable tissues that, when fully developed, can occupy up to 12% of the plant cross sectional area (Armstrong 1978).

Oxygen transport in wetland plants was originally thought to be solely a passive diffusion process driven by oxygen concentration differences between the root zone and the external atmosphere (Armstrong 1978). Later work has shown that oxygen transport is also driven by light, temperature, and humidity differences between the

internal plant tissues and the outside environment. This mechanism causes an internal pressure differential that in effect pumps oxygen downward and results in an active convective flow of gases to the roots (Armstrong and Armstrong 1988,1990,1991; Armstrong et al. 1990; Brix 1988).

To be of use in the root zone for nitrification, the oxygen has to be made available to nitrifying microorganisms. It is believed that oxygen leaks from the roots and creates aerobic microzones in which nitrification can take place. As discussed earlier, this oxygen leakage is evidenced by the oxidation of phytotoxins and the oxidized forms of iron and manganese precipitated onto the roots of wetland plants.

A great deal of work on oxygen transfer in aquatic plants is being accomplished in Europe. Most of the work is centering around Phragmites australis (common reed) because of its extensive internal oxygen pathways. The stem (culm) is almost hollow, providing an almost clear pathway for oxygen to be transported into the roots of the plant. Reeds are an ubiquitous species that form dense nearly monospecific stands bordering streams, lakes, and even brackish water areas in Europe (Brix 1988). The reed also is the plant used almost exclusively in Europe for SFCW and the primary reason for its extensive study.

This section will discuss the internal oxygen transport mechanisms of wetland plants (primarily reeds) and the availability of oxygen in the root zone of wetland plants for waste treatment.

B. Diffusion

Diffusion was originally thought to be the only mechanism for oxygen transport in aquatic plants. Molecular oxygen enters through the aerial parts of the plant and moves by gas phase diffusion via the intercellular void spaces to the submerged roots (Armstrong 1978). The diffusion of air by this mechanism is driven solely by the concentration differences between atmospheric oxygen and the oxygen available in the plant roots. In its simplest form, the mass flux of oxygen can be modeled by:

$$\text{Mass Flux (g/s)} = - DA\,(C_2 - C_1)/L$$

Where D is the diffusion coefficient (cm^2/s), A is the cross sectional area of the the plant section (cm^2), C_2 is atmospheric oxygen content and C_1 is the oxygen concentration in the roots (g/cm^3), and L is the diffusion path length (cm). In reality, the void spaces are distributed nonuniformly and can be significantly tortuous to effectively lengthen the diffusion path quite considerably. The model is further complicated by nonuniform respiratory requirements along the submerged root and oxygen leakage from the roots into the surrounding soils (Armstrong 1978). This is a very simplistic approximation of a complex phenomena, but does present a good intuitive description of the mechanism.

Brix and Schierup (1990) attempted to quantify the amount of oxygen that can diffuse by this mechanism by observing the oxygen transfer through the dead hollow culms of reeds in a SFCW located in Denmark during winter. In winter the aerial parts of the plants die leaving a stem (culm), but the dead culm still provides a pathway for oxygen to be transported to the still active roots and rhizomes. Recognizing that oxygen transport mechanisms not only include diffusion but temperature and humidity driven convective transport as well, it was thought that measuring the oxygen transport in the winter would best approximate the passive diffusion (Brix 1989). Brix and Schierup (1990) measured 2.08 g O_2/m^2-d transferred through the standing dead reeds. They also measured the amount of oxygen utilized by the below ground roots and rhizomes and concluded that the below ground plant tissues were utilizing approximately 2.06 g O_2/m^2-d, leaving potentially only 0.02 g O_2/m^2-d to be released to the surrounding soil. They concluded that passive diffusion alone is not sufficient enough to provide the oxygen required for aerobic BOD removal and nitrification in SFCW. This conclusion helps explain the poor performance of these systems in Denmark and other northern climate locations.

C. Convective Oxygen Transport

As complicated as passive diffusion is, convective transport is more so and even less understood. Experimental results do support the existence of convective flow and show a definite relationship between light, temperature and humidity (Armstrong and Armstrong 1988,1990,1991; Armstrong et al. 1990; Brix 1988).

Brix (1988) experimented with reeds by measuring the composition of the internal air (CO_2, O_2, and N_2) during alternating light and dark cycles. Table IV-1 summarizes his results and indicates a steep concentration gradient from the aerial parts of the plants to the deepest growing rhizomes and a clear relationship between light and dark internal gaseous concentrations. He attributed the increase in the amount of gas flow to thermo-osmotic Knudsen pressurization resulting from temperature differences between the aerial plant tissues and the surrounding air. Brix also concluded that because the O_2 consumed should be replaced by approximately an equal volume of CO_2 (his results indicated that this was not occurring) and the observation that concentrations of dissolved inorganic carbon increased in the root zone, that respiratory CO_2 was being released into the root zone. This release results in a decrease in internal pressures in the rhizome and an increase in the pressure gradient downward, facilitating oxygen transport.

Armstrong and Armstrong (1991), through a series of experiments with reeds, were able expand on the relationship of light to internal oxygen transport as well as the effects of temperature and humidity. Their first experiment was conducted under low light conditions (20 μmole/m^2-s photosynthetically active radiation (PAR)) to isolate the relationship between temperature and humidity. As can be seen in Figure IV-1, their data shows that increasing the atmospheric humidity greatly reduced convective flow rates. At 24^o C, the flow at 41% relative humidity was 3.5 times the rate at 74 %. Convective flow rate also increased with increasing air temperature as seen in Figure IV-2.

In another experiment, by sealing portions of the plant in attempt to stop gases from flowing they were able to establish static pressure differentials of 200 Pa in the internal plant structure under moderate light conditions (180 μmole/m^2-s PAR) and pressures 4 times larger under intense light (1000 μmole/m^2-s PAR) . Also under intense light, convective flow rates of 0.0324 x 10^{-6} m^3/s were recorded which were 8.5 times that recorded at low light intensity. Their work showed a definite relationship between light, temperature, and humidity. Finally in an attempt to locate from what part of the plant the air was flowing, the leaves and aerial structures were covered with a "cling film". At a temperature of 20^o C, 47% humidity, and a light flux of 180 μmole/m^2-s, convective air flow was reduced from 0.0097 x 10^{-6} to 0.0053 x 10^{-6} m^3/s when the cling film was applied.

Their data indicated that convective flow is caused by an inflow of gases through the pore structure in the leaves and that larger leaves allowed larger volumes of air to be moved. Humidity differences between the inside and the outside of the plants also contributes to the flow of gases. Armstrong and Armstrong (1991) postulated that the internal plant humidity was higher than the outside humidity and the higher humidity on the inside had a tendency to displace some of the oxygen creating an inward diffusion of air from the drier air on the outside. They further suggested that the internal humidity was maintained by the evaporation of water from the internal cell lining of the internal void spaces.

Light appears to be the primary driving mechanism for this process. Light can increase the temperature within the plant by as much as 4^o C (Armstrong and Armstrong 1991). This increase in temperature may produce conditions that favor the in flow of oxygen by thermally induced Knudsen diffusion and also may increase the internal humidity such that oxygen is displaced causing an even higher oxygen gradient. Light may also affect the mechanism by influencing how the leaves external pore

structure responds. Armstrong and Armstrong (1991) suggest that the light may cause the pores to open up allowing more atmospheric gas to enter.

The experimental data do support that pressurized convective flow is occurring and researchers (Armstrong and Armstrong 1988,1990,1991; Armstrong et al. 1990) have attempted to explain the mechanisms driving it, but there is still great uncertainty. For example, the mechanisms that are creating gaseous inflow gradients are also creating gradients that would tend to force gases out. Both the temperature and humidity gradients are in the opposite direction, i.e., out of the plant. Also, the aerobic respiration in the roots is creating by products (CO_2), and, as Brix (1988) showed, the gradient is out of the plant. The leaves pore structure must be such that it allows Knudsen diffusion of oxygen into the plant and selectively allows the expulsion of carbon dioxide out of the plant. What is causing and controlling the convective flow still is not fully understood.

D. Root Zone Oxygenation

The evidence is fairly conclusive that oxygen is moving into the roots of aquatic plants by a combination of diffusion and convection. The next questions that need to be addressed are how does the oxygen move into the surrounding soil (root zone) and how much is actually being made available for root zone aerobic processes such as nitrification.

Armstrong and Armstrong (1988) working with reeds, studied how oxygen moves from the roots into the root zone. By utilizing methylene-blue oxidation and various root structures, they were able to show how the oxygen moves out of the root structures. They found that oxygen release was most rapid from young adventitous and secondary roots, particularly basal tufts of fine laterals. They also found zones of oxidation around sprouting tips of both vertical and horizontal rhizomes. They found little oxygen transfer from older rhizome tips and the rhizomes themselves. Maximum

zones of oxidation extended 8 mm after 24 hours from growing horizontal rhizomes and 10 mm for basal laterals after 1.5 hours.

Several authors (Armstrong and Armstrong 1991, Reed et al. 1992, and Reddy et al. 1989) have made attempts to quantify the amount of oxygen being released to the root zones of aquatic plants. Armstrong and Armstrong (1991) measured the oxygen flow from a typical reed plant in the laboratory at 20° C and moderate light intensity and estimated that a wetland bed of reeds would provide between 5-12 g O_2/m^2-d.

Reddy et al. (1989) using a variety of plants placed singly into 500 mL flasks of primary sewage and sealed against the intrusion of outside air calculated the oxygen transport of individual plants by measuring the BOD, DO, TKN, NH_4-N, and (NO_3 + NO_2)-N before and after 8 days at 25° C with a 12-h light /12-h dark cycle. They then calculated oxygen transport rates by doing an oxygen mass balance, their data is seen below.

Plant	mg O_2/d
Pontederia cordata	12.5
Common cattail	4.4
Common arrowhead	8.3
Canna flaccida	7.9
Scirpus pungens	3.4
Scirpus validus	7.3

Reed (1992) by doing an oxygen mass balance using Gersberg's et al (1986) performance data calculated that reeds would transport 4.8 g-O_2/m^2-d, bulrush (Scirpus validus) 5.7 g-O_2/m^2-d, and cattails 2.1 g-O_2/m^2-d.

F. Discussion

All of the discussed work on oxygen transport was undertaken not only to gain an understanding of the oxygen transport mechanisms in in aquatic plants but was also undertaken in the context of wastewater treatment. Armstrong and Armstrong, Brix, and Schierup are aquatic botanists who have done a great deal of work with emergent aquatic plants and subsurface flow constructed wetlands. As discussed earlier in this paper, the

key to sequential nitrification/denitrification is oxygen transport into the root zone, and, for proper design, the quantification of the oxygen transported.

Armstrong and Armstrong's (1988) work also supported the requirement discussed earlier for the design of SFCW, that to be effective the waste must come in contact with the root zone and bed depth must match root penetration. Not all of the root structures release oxygen, and that which is released does not diffuse very far away from the roots (< 10 mm).

There is a great deal of similarity between the oxygen transport numbers obtained experimentally by Armstrong and Armstrong (1991) and those obtained by Reed (1992) by mass balance calculations. It is possible that these numbers can be used for design criteria as previously discussed and discussed further in the following sections.

An understanding of the process is developing as well as other factors that may influence (light, temperature, and humidity) oxygen transport, but more work is needed.

Table IV-1. Mean composition (%) of the air in different parts of reeds during light and dark cycle.

Plant Part*	O_2 Light	O_2 Dark	CO_2 Light	CO_2 Dark
Stems(50-80 cm)	20.7%	20.5%	0.07%	0.2%
Stems(0-20 cm)	20.7%	20.2%	0.09%	0.32%
Roots (0-20 cm)	18.9%	14.5%	1.11%	2.73%
Roots(20-40 cm)	14.9%	9.9%	3.49%	5.04%
Roots(40-60 cm)	12.5%	6.3%	4.6%	6.39%
Roots (60-80 cm)	10.4%	7.8%	6.32%	6.71%
New Rhizomes(50-80cm)	5.3%	3.6%	7.31%	7.35%

(After Brix 1988).* Elevation above or below the ground surface in cm.

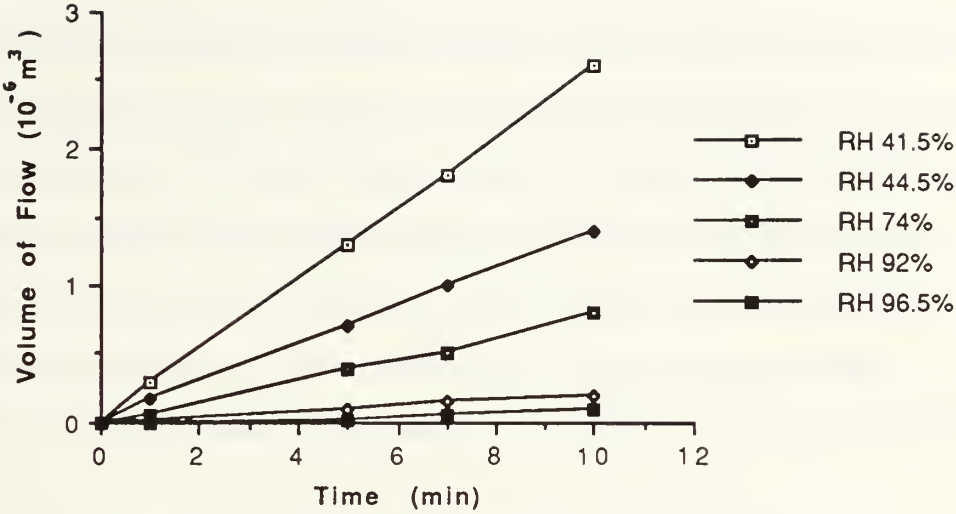


Figure IV-1. Relationship between relative humidity and convective flow (Armstrong and Armstrong 1991).

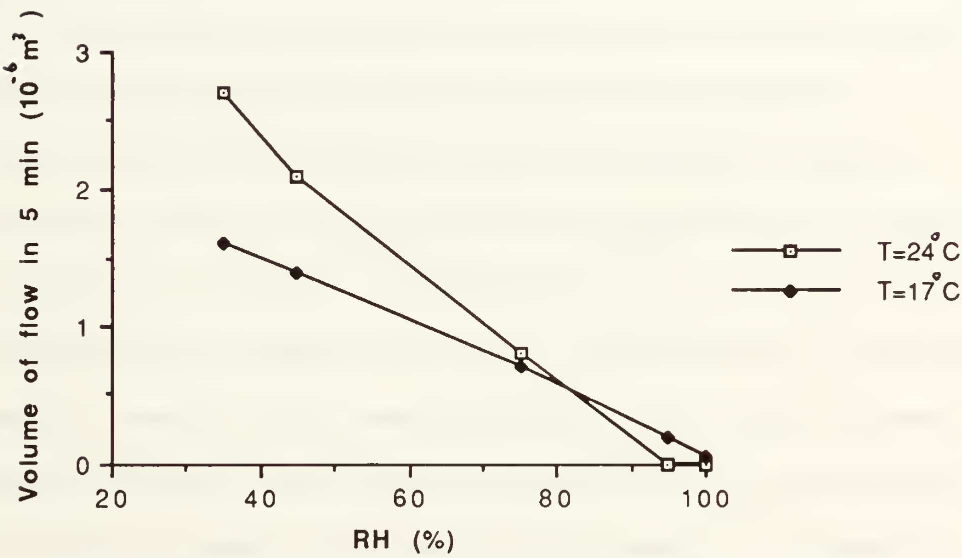


Figure IV-2. Relationship between temperature and convective flow. (Armstrong and Armstrong 1991)

SECTION V. DESIGN RECOMMENDATIONS

A. Introduction

As previously discussed, SFCW were initially embraced with great enthusiasm as an effective wastewater treatment alternative, but in practice they have failed to live up to expectations. Removals of BOD and TSS have generally been good, but nitrogen removal can be rated poor at best. The potential exists to utilize SFCW as a nitrogen removal option, but more understanding of the process is required before they can be designed consistently and with confidence. On the positive side, there are a few successful systems, and these seem to have some common characteristics which will be discussed later.

Part of the problem is poor hydraulic design resulting in surface flow. As discussed in Section II, this problem can be alleviated by improving construction practices, using a safety factor of two in Darcy's law when calculating bed cross sectional area or flow, and installing adjustable outlet structures to improve control of water depth in the beds. The biggest problem stems from there being almost no "rational" design approach for sequential nitrification/denitrification. "Rational" defined here means an approach or equation based on an understanding of what is going on physically, chemically, or biologically. The EPA (1988) offers only a recommendation of 5-7 days detention time to produce an effluent of 10 mg/l NH_3-N, with no distinction between primary or secondary influent loadings or any other explanation. European design guidelines (EC/EWPCA 1990) recommend not using SFCW for nitrogen removal. The WPCF (1990) offers a design equation (see Section II) based on a regression of "selected" data that includes surface and subsurface flow as well as natural wetlands. Both the EPA and WPCF approach offer very conservative designs, and, based on past performance of most systems, the more conservative the better. There are systems, though, that produce effluents within secondary standards and with HRT as low as 24 hours (Jones 1992). Reed et. al. (1992) is one of the only attempts to offer a rational

design approach by basing the size of the wetland on an estimate of the oxygenation capacity of the wetland plants. This approach needs to be used with caution as the oxygenation capacity is very site dependent, being governed by the environment (light, temperature, and humidity).

The biggest obstacle to developing a rational design approach is the lack of performance data. Most of the data in the literature consist of nothing more than average flow, surface area, and influent and effluent concentrations. As will be shown later, most of the systems do not perform very well, therefore, there are very few conclusions that can be drawn or relationships that can be determined. Most of the available data are from the United Kingdom and Denmark with very little performance data available from operating systems in the United States. The EPA (Reed and Brown 1992) has completed an inventory of the wetland systems in the U.S., but the inventory has not reached the point where performance data has been collected and are available for analysis. Though data are limited, some detailed performance reports from some systems were located, and these proved to be very helpful (Watson et al. 1990 and TVA 1990).

This section presents and discusses the available performance data and offers a design approach (originally proposed by Bavor 1988) developed from the few systems that did produce effluents within secondary standards.

B. Performance Data Analysis

Performance data was compiled (see Appendix A) from 43 SFCW located in the United Kingdom, Denmark, and the United States. The data was collected from journal articles, conference proceedings, and government reports. Some sites reported nitrogen as total nitrogen, some as ammonia nitrogen, and some as both.

Looking at the data, it becomes readily apparent that only a few systems, six, were capable of producing an effluent within secondary standards. No sites had any data on hydraulic conductivity of bed media, and only three had any data on hydraulic retention times. Only five sites had any data on root penetration of the bed. It is

interesting to note that of all the sites only three reported full root penetration, and all of these produced an effluent within secondary standards.

Hydraulic and nitrogen loading versus performance was analyzed to look for possible relationships and to develop loading factor recommendations for design. Most of the data presented in terms of TN was from Denmark and most of these systems performed poorly. As can be seen in Figures V-1 and V-2 performance was extremely poor and inconsistent overall. It was decided that data analysis in terms of TN would be inconclusive and produce no dependable loading criteria.

The performance data in terms of ammonia was not much better, but some patterns were seen that could be interpreted and lead to reasonable estimates for loading criteria. As can be seen in Figures V-3, V-4, and V-5, there are six constructed wetlands that produced effluents within secondary standards and had removals greater than 50%. Of the six, three were known from the literature to have achieved full bed penetration of their root zones. Based on these SFCW, it is recommended that the HLR be kept under 4 cm/day and ammonia loadings under 10 kg/ha-d. The EPA and the WPCF have no recommendations for ammonia loading but recommend a HLR of 2 and 2.5 cm/day for maximum treatment efficiency. A 4 cm/day hydraulic loading is higher than recommended by EPA and WPCF, but their numbers were based on systems that did not achieve full root penetration of the wetland beds. Bringing all the influent in contact with the root zone should allow for greater hydraulic loading.

C. The Plug Flow First Order Model

The design of SFCW for BOD removal is based upon a first order plug flow model, and this model should hold true for ammonia removal as well. The equation is as follows:

$$A_s\ (m^2) = Q(\ln NH_3in/NH_3eff)/(K_T dn)$$

where:

Q = flow rate (m^3/s)
NH_3in = concentration of influent (mg/l)
NH_3eff = concentration of effluent (mg/l)
d = depth of liquid in bed (m)
n = bed porosity

K_T = ammonia rate removal constant (day^{-1}).

K_T (day^{-1}) is the temperature dependent rate removal constant. Bavor (1988) using the first order plug flow model for a SFCW in Australia, calculated an ammonia rate removal constant 0.107 day^{-1} (at T=20^o C with $K_T = K_{20}(1.03)^{T-20}$). His number is based upon a system that did not perform very well as a result of poor bed oxygenation. Table V-1 shows the calculated rate removal constants for the six SFCW systems that performed well in this analysis. The literature for these sites did not include any temperature data so these rate constants could not be tied to temperatures. The hydraulic retention time for Middleton and Bear Creek was calculated based upon porosity estimates of 0.32 for a sand bed and 0.40 for a gravel bed. The average rate removal constant of the six values is 0.364 day^{-1}, this value on average represents a broad temperature range as the sites are very diverse in terms of location. Reddy et al. (1989), for a variety of aquatic plants, calculated an ammonia rate removal constant range of 0.057 - 0.257 day^{-1}. Reddy et al. (1989) unfortunately did not tie the rate constants to a particular plant.

Continuing the example started in Section III with:

$$K_T = 0.364 \ day^{-1}$$
$$n = 0.4$$
$$d = 23 \ cm$$

$$As(m^2) = (75.7)(\ln 20/8)/((.364)(.4)(.23)) = 2080 \ m^2$$

Thus, the required area is As=0.21 ha or 22,400 ft^2. The area requirements calculated from the two approaches presented in Section III and the plug flow first order model are shown below:

Design Approach Comparison

Design Approach	Area (ha)
WPCF	0.3
Reed et al. (1992)	0.22
Plug Flow Model	0.21

The Reed et al. (1992) approach and the plug flow model approach produced almost identical results. Both are based on what is actually going on biologically in the SFCW. Reed's et al. approach is based on possible oxygen transfer rates, and the plug flow model approach is based upon biological rate constants that are dependent upon oxygen transfer as well.

D. Design Recommendations

i. Primary or Secondary Effluent

When a SFCW is being designed for a primary influent and nitrogen removal is desired, the area for BOD and TSS removal is determined first. The wetland should be designed to lower the BOD to about 20 mg/L (Reed et al. 1992) to ensure sufficient carbon for denitrification but not allow too high a carbon to nitrogen ratio to inhibit the growth of denitrifying organisms. After the area for BOD removal is determined, the area for sequential nitrification/denitrification should be calculated utilizing one of the approaches outlined in Table V-2 and added onto the total SFCW area. If the bed is being designed as an add-on system for nitrogen/ammonia removal to polish secondary effluent, the upstream processes (oxidation ponds, aerated lagoons, activated sludge, etc.) should be adjusted to deliver a wetland influent of about 20 mg/l BOD and then the SFCW designed accordingly (Reed et al. 1992).

ii. Bed Media

It is recommended that gravel or course sand be considered as the bed media to ensure that subsurface flow is maintained. Table V-2 shows typical characteristics for various bed media. There is a tradeoff between smaller media size and increased surface area for microorganisms, and larger media and increased hydraulic conductivity allowing for smaller beds. Fine to medium gravel is the best choice as sands will tend to clog over time reducing the efficiency of the bed. It is also recommended that actual material porosity and hydraulic conductivity be determined for final designs. Construction practices need to be monitored to ensure that the bed is not contaminated

with a large amount of fines during media transportation and installation (Reed et al. 1992).

iii. Vegetation and Bed Depth

To ensure high treatment efficiency, it is vital that the wastewater come in contact with the root zone to take advantage of the oxygenation capacity of the aquatic plants. As discussed previously, Gersberg et al. (1986) measured root lengths for some typical SFCW aquatic plants (Table V-2), and these are thought to be about the maximum root lengths attainable. Although these plants are the most common grown in SFCW, they are not the only plants that can be considered. For example swamp potato (Sagittaria latifola) and duck potato (Sagittaria falcata) are being used with good results (Zachritz and Fuller unpublished data) for BOD removal, and Thaylia (T. dealbata and T. divericata) is obtaining outstanding results for both BOD and ammonia removal (Jones 1992).

iv. Bed Area

Bed surface area can be determined one of three ways. As seen in Table V-2, the WPCF design equation can be utilized, the plug flow first order model equation can be utilized, or the surface area can be calculated by a mass balance between the amount of oxygen required to nitrify a given loading of ammonia and the oxygen transport capability per unit bed surface area of wetland plants.

No recommendation can be given on what approach to use for calculating required bed area. If confidence is high that plant oxygenation capacity or the ammonia rate constant can be adequately estimated, the oxygen mass balance or the plug flow approach will be the best for calculating required bed areas. If these critical parameters cannot be estimated, then the more conservative WPCF equation should be used.

v. Hydraulics

To ensure subsurface flow, as discussed in Section II, construction procedures (e.g., tarps on trucks and dust control), need to be explicitly stated in the construction

specifications to avoid contamination of the bed media with fines. Darcy's law can be used for bed design, but a factor of safety of two should be incorporated into the hydraulic conductivity to ensure subsurface flow. Finally, it is recommended that adjustable outlet structures be installed so that flow through the bed can be more precisely controlled (Reed et al. 1992).

v. Multiple Beds

The flow should be distributed over several parallel beds to ensure operational flexibility for maintenance and to allow for periodic wastewater drawdown to ensure root penetration. Root penetration cannot be left to chance, and constantly saturated beds will result in shallow root structures. To date this has not been practiced routinely in the U.S., but is recommended in the European design manual for SFCW (Reed et al. 1992).

E. Design Example

Assume a small town near the Gulf coast with an average flow of 1000 m^3/d. The town has two aerated lagoons in series that meets its discharge requirements for BOD_5 (20 mg/L) and TSS (20 mg/L) but cannot meet its requirements for ammonia. The system is currently discharging an effluent of 25 mg/L NH_3-N. The town wants to construct a SFCW to provide an effluent of 5 mg/L NH_3-N.

<u>Step 1 - Bed Media</u>

Select a medium gravel bed media with a porosity of n=O.4 and a hydraulic conductivity of ks=10,000 m^3/m^2-d.

<u>Step 2 - Vegetation and Bed Depth</u>

Choose <u>Phragmites</u> <u>australis</u> (reed) as the vegetation and a bed depth of 50 cm.

<u>Step 3 - Calculate Required Bed Surface Area</u>

All three approaches will be calculated for comparison. Since the town is located in an area that is known for its warm sunny days all year round, the ammonia rate

removal constant is assumed to equal 0.35 day^{-1} and the oxygen transfer rate is 4.5 g/m^2-d.

WPCF---> $As = (100)(1000)/\{\exp[(1.527)(\ln 5)-(1.05)(\ln 25)+1.69]\}=46,404\ m^2$

Plug Flow Model---> $As = (1000)(\ln 25/5)/[(.35)(.5)(.4)]=22,992\ m^2$

Mass Balance---> $As = (25\text{-}5)mg/l(1000m^3/d)(5mgO_2/mgNH_3)/4.5=22,222\ m^2$

Assume because of the ideal weather conditions that 23,000 m^2 of SFCW surface area will be required.

Step 4 - Aspect Ratio

As discussed in Section II, an aspect ratio of 0.4 to 3:1 is recommended to ensure sufficient a hydraulic gradient through the bed. This is a good recommendation, but needs to be balanced against the need to maintain plug flow. Assume a length:width ratio of 3:1 to ensure plug flow, but not restrict hydraulic gradients through the bed and the range of adjustments on the outlet devices.

Step 5 - Bed Cross Sectional Area

Calculate the cross sectional area of the bed using Darcy's law. Assume a bed slope of 1% and a hydraulic conductivity of 5,000 m^3/m^2-d after a factor of safety of 50% (see Section II) is applied.

Cross sectional area = $Q/(ks \times S) = (1000)/[(5000)(.01)] = 20\ m^2$

Step 6 - Bed Dimensions

$W = 20/.5 = 40$ m

$L = 40 \times 30 = 120$ m

Step 7 - Number of Beds

No. of beds = 23000/4800 = 4.8 beds

Five beds plus one additional for operational flexibility for a total of six.

Step 8 - Hydraulic and Ammonia Loading

HLR = 4.2 cm/d

Ammonia Loading = 10.4 kg/ha-d

As discussed earlier, a HLR of 4 cm/d and ammonia loading of 10 kg/ha-d maximum is recommended (this author). The bed surface area possibly should be adjusted to lower both the HLR to below 4 cm/d and the ammonia loading to below 10 kg/ha-d.

F. Discussion

Overall, the most difficult to determine and most important design consideration for these systems is the required surface area. Designers have to make a decision whether to use the non-rational conservative design approach recommended by the WPCF, or a more rational approach that may be based on limited amounts of data. As can be seen in the above example, the approach chosen can result in a system that varies from the alternative by a factor of two.

Table V-1. Selected SFCW performance data.

Site	Q (m^3/d)	NH_3in (mg/l)	NH_3eff (mg/l)	HRT (days)	K_T
Marnhull #1	85	28	10	4.6	0.224
Marnhull #2	85	28	9	4.6	0.246
Middleton	35	2.3	0.8	2.47	0.427
Santee #1	3.05	24.7	5.3	6	0.256
Santee #2	3.05	24.7	1.4	6	0.478
Bear Creek	57.54	10.7	1.8	3.23	0.552

Table V-2. SFCW design considerations.

Design Consideration	Options	Characteristics		
Bed Media		D_{10}(mm)	n*	ks*
	Coarse Sand	2	0.32	1000
	Gravelly Sand	8	0.35	5000
	Fine Gravel	16	0.38	7500
	Medium Gravel	32	0.40	10000
	Coarse Rock	128	0.45	100000
Vegetation		Root Depth(cm)		O_2(g/m^2-d)
	Bulrush (Scirpus)	76		5.7
	Reed (Phragmites)	60		4.8
	Cattails (Typha)	30		2.1
Bed Area (m^2)	WPCF (1990)	=100Q/{exp[(1.527lnCe)-(1.050Co)+1.69]}		
	Reed (1992)	=O_2required(g/d)(O_2 (g/m^2-d))		
	Plug Flow Model	= Q(ln Co/Ce)/($K_T$dn)		
K_T (Ammonia Rate Removal Constant)	Bavor (1988)	K_{20} = .107 d^{-1}		
	Titus (1992)	K_T = .364 d^{-1}		
	Reddy et al. (1989)	K_T = .057 - .252 d^{-1}		
L/W Ratio		0.4 :1 to 4:1		
Hydraulics	Darcy's Law	Q = ksAS(dh/dl)		
HLR	EPA (1988) WPCF (1990) This Report	≤ 2 cm/d ≤ 2.5 cm/d ≤ 4 cm/d		
Ammonia Loading	This Report	10 kg/ha-d		
* Note n=porosity ks=hydraulic conductivity (m^3/m^2-d)				

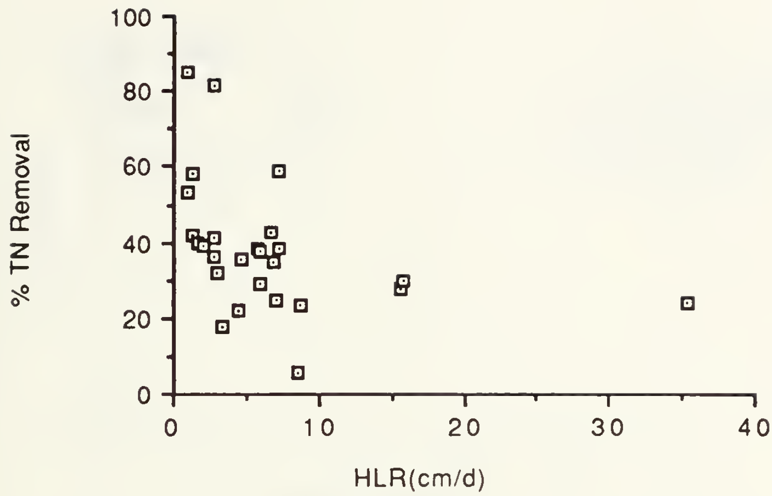


Figure V-1. HLR vs. %TN Removal. Figure shows poor performance and inconsistent data.

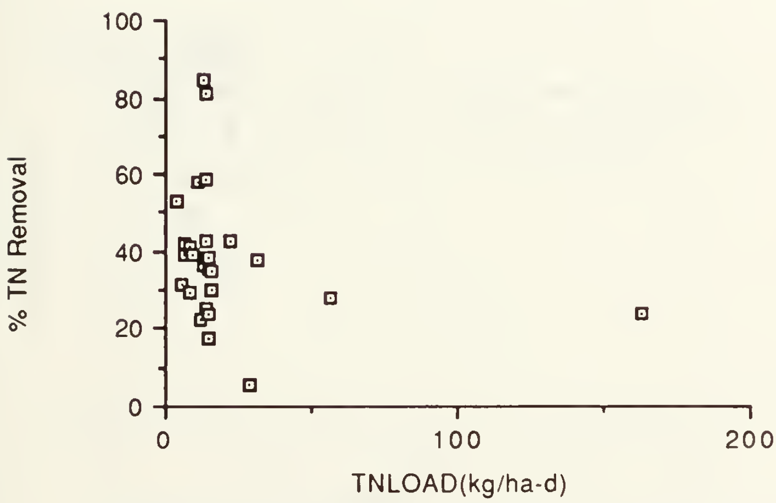


Figure V-2. TNLoad vs. %TNRemoval. Figure shows poor performance and inconsistent data.

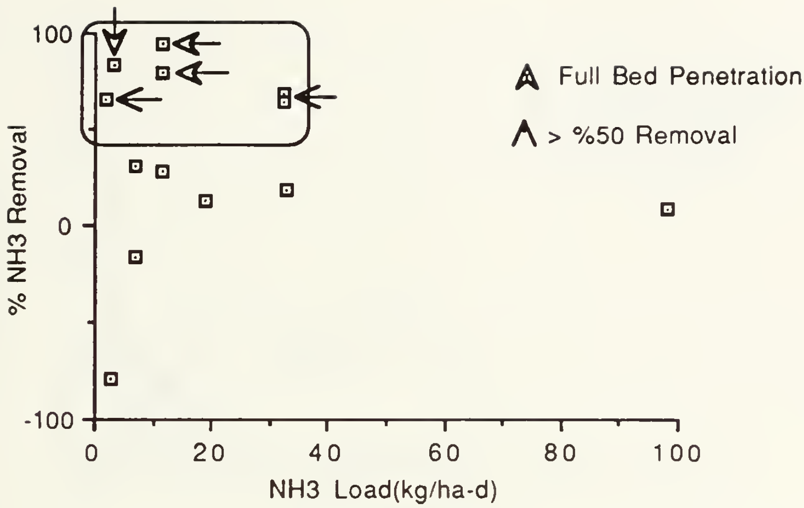


Figure V-3. NH_3 Load vs. % NH_3 Removal. Bracketed points are the six systems out of 43 reviewed that performed well.

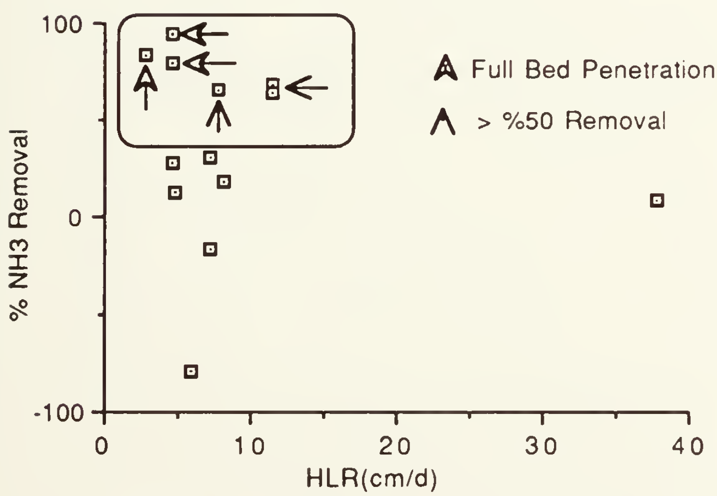
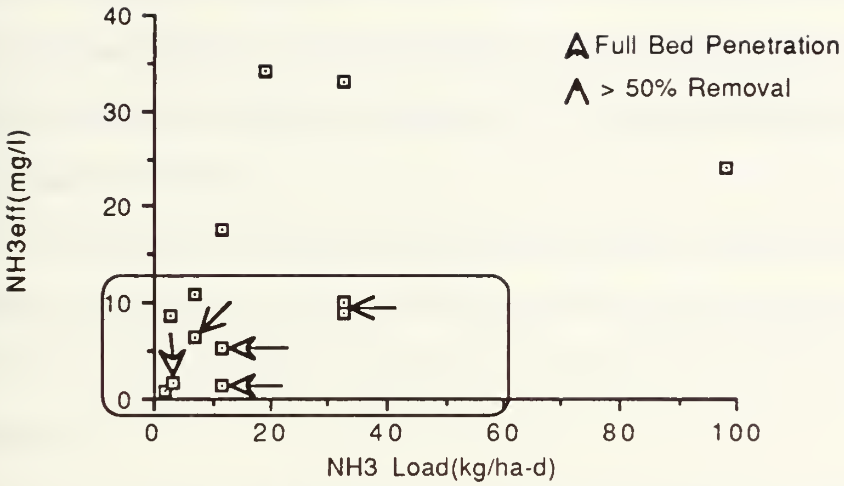


Figure V-4. HLR vs. % NH_3 Removal. Bracketed points are the six systems out of 43 reviewed that performed well.



<u>Figure V-5</u>. NH_3 Load vs NH_3 Effluent. Bracketed points are the six systems out of 43 reviewed that performed well.

VI CONCLUSIONS AND RECOMMENDATIONS

A. Summary of Objectives

Subsurface flow constructed wetlands are a potentially good treatment option for small communities looking for a low maintenance, low technology system. The applicability of SFCW technology has been shown by meeting the objectives of this report, restated from Section I:

1. To discuss the development of SFCW technology, review the variety of design approaches, and discuss the overall performance of these constructed wetlands;

2. To discuss nitrification/denitrification in SFCW and typical problems encountered;

3. To discuss emergent aquatic plants and oxygen transfer (the key to sequential nitrification/denitrification);

4. To review nitrification/denitrification performance data and develop recommendations on hydraulic loading, retention times, and nitrogen loading; and

5. To offer conclusions on the further applicability of SFCW technology and recommendations on further research needs.

B. Conclusions

i. SFCW and BOD/TSS Removal

The use of the first order plug flow model for BOD removal appears to be a good design approach for these systems and does produce effluents within secondary standards. Most current systems experience surface flow resulting in short circuiting. The problem seems to be caused by poor hydraulic design, resulting from over estimation of the bed media's hydraulic conductivity and poor construction practice. Poor construction practice results in the introduction of fines into the media, which decreases the conductivity. The use of Darcy's law is appropriate for the design of these systems, but it is recommended that a safety factor of two be used for the hydraulic conductivity to ensure subsurface flow.

ii. SFCW and Sequential Nitrification/Denitrification

SFCW for sequential nitrification/denitrification also shows promise, but currently most systems are not performing very well. Of the 43 systems reviewed for this report, only six systems were able to produce an effluent within secondary standards. The major problem appears to be insufficient oxygenation of the root zone, resulting in poor nitrification. Wetland plants have the ability to transport oxygen to their roots, and leaking oxygen from the roots to the root zone results in aerobic microzones that allow nitrification to occur in this otherwise anoxic environment. Oxygen leaking from the roots does not diffuse very far, about 10 mm maximum, therefore, the wastewater must come in contact with the root zone and bed depth must be matched to root zone depth. Different wetland plants have different root growth potential, so the selection of a particular wetland plant determines the bed depth.

Designers currently have three design approaches: the WPCF regression analysis, the first order plug flow model approach, and the oxygen mass balance approach, resulting in the design of SFCW for nitrification/denitrification, at this point, being more of an art than a science. The conservative WPCF approach results in SFCW systems that are almost twice as large as those resulting from the more rational plug flow model and oxygen mass balance approach. The WPCF approach is based on the performance of a number of different wetland systems (both surface and subsurface) that performed marginally at best (Reed and Brown 1992), and the other two approaches are based on a small number of subsurfaces systems that preformed extremely well. The data that is available from the good performing systems is incomplete and sketchy, and the derived rate removal constants, in this report, for the plug flow model and the oxygen transport estimates for the mass balance approach (Reed et al. 1992, Armstrong et al. 1990) are based on a very limited amount of data. No recommendation can be given as to which approach is best as there is currently insufficient data from which to draw a conclusion. Reed and Brown (1992) describes the

design of SFCW as being "seat of the pants" design and the choices presented by the different design approaches show this to be true.

C. Recommendations

Oxygen transport by aquatic plants is the most important mechanism involved in sequential nitrification/denitrification of wastewaters in SFCW. Most of the U.S. literature reviewed limits the discussion of oxygen transport to only stating that wetland plants have the ability to transport oxygen to their root zone, and no more. There is no mention of what is driving this mechanism and its dependence on light, temperature, and humidity, which possibly indicates that designers do not understand what is going on inside the wetland and the roles of different aquatic plants. The EPA Process Design Manual: Land Treatment of Municipal Wastewater (EPA 1981) emphasizes the need for designers to work closely with soil and plant scientists, to ensure that correct design decisions are made with regards to these highly specialized areas. The same recommendations do not exist in the current EPA (1988) and the WPCF (1990) design guidance for SFCW. Designers of SFCW need to work closely with aquatic botanists and wetland ecosystem experts, for guidance in plant selection and operation and maintenance procedures to ensure optimal performance of SFCW. Designers also need to keep up with the current on-going work attempting to quantify the oxygen transport capacity of wetland plants. Understanding this mechanism is key to the proper design of these systems.

To adequately quantify nitrogen rate removal constants and oxygen mass transfer, detailed performance data from individual systems is needed to show the relationship between performance and operating conditions. The U.S. EPA Risk Reduction Environmental Laboratory in Cincinnati, Ohio is attempting to accomplish this by studying individual SFCW systems in detail (Reed and Brown 1992). The studies are a beginning, but are limited to the summer months, which represents a very narrow range of operating conditions. Also, the sites selected, Mandeville and Carville Louisiana,

did a poor job in denitrifying wastewaters (Reed 1991) and the data collected is of limited use for studying the nitrification/denitrification process.

What is needed is a continuation of the Gersberg et al. (1986) work with SFCW. A rationally designed system needs to be constructed and placed in a geographic area with some seasonal variation or in an environmentally controlled greenhouse. The SFCW should then be studied and the relationship between removal efficiencies and light, temperature, and humidity determined. This study will allow rate removal constants to be determined, for both BOD and nitrogen, and better estimates of oxygen transfer made. More can be learned from studying one system in detail, one that is designed well and performs well, than can be learned studying a large number of systems superficially that perform marginally.

Recently, some of the literature has recommended that SFCW not be designed for sequential nitrification/denitrification and only for denitrification, an anoxic process. Reed and Brown (1992) point out that some designers are including open water or overland flow into their treatment systems before introduction into a SFCW, to promote aeration and thus nitrification. This recommendation may be necessary based on the current status of the design procedures, but as more information becomes available on SFCW and the process of aquatic plant oxygen transfer, designers will be able to design systems that take full advantage of aerobic and anaerobic microzones and aquatic plants as natural oxygen pumps.

APPENDIX

APPENDIX A-1
SFCW GENERAL CHARACTERISTICS

LOCATION		REF.	TYPE	VEGETATION	AREA (m^2)	Q (m^3/d)	HLR (cm/d)	L:W Ratio	DEPTH (cm)	ROOT ZONE DEPTH(cm)
Benton#3	KY	1	Gravel(S)	Bulrushes	14652.0	863.0	5.9	7.6:1	61	7
Hardin#1	KY	1	Gravel(S)	Reeds	3190.0	230.0	7.2	6.6:1	61	-
Hardin#2	KY	1	Gravel(S)	Bulrushes	3190.0	230.0	7.2	6.6:1	61	-
Gravesend#1	UK	2	Gravel(P)	Reeds	1225.0	100.0	8.2	1:01	60	-
Gravesend#2	UK	2	Gravel(P)	Reeds	1225.0	100.0	8.2	1:01	60	-
Gravesend#3	UK	2	Gravel(P)	Reeds	1225.0	100.0	8.2	1:01	60	-
Holtby	UK	2	Soil(P)	Reeds	612.0	30.0	4.9	0.52:1	60	-
Marnhull#1	UK	2	Soil(P)	Reeds	735.0	85.0	11.6	0.6:1	60	-
Marnhull#2	UK	2	Soil(P)	Reeds	735.0	85.0	11.6	0.6:1	60	-
Middleton	UK	2	Sand(S)	Reeds	450.0	35.0	7.8	0.5:1	60	-
Valley Field	UK	2	Gravel(P)	Reeds	45.0	17.0	37.8	0.45:1	60	-
Bluther#1	UK	3	-	Reeds	40.0	4.3	10.6	0.45:1	60	-
Bluther#2	UK	3	-	Reeds	40.0	4.3	10.6	0.45:1	60	-
Bluther#3	UK	3	-	Reeds	40.0	4.3	10.6	0.45:1	60	-
Bluther#4	UK	3	-	Reeds	40.0	4.3	10.6	0.45:1	60	-
Castleroe	UK	3	-	Reeds	30.0	5.0	16.7	2.2:1	60	-
Moesgard	DK	4	Soil	Reeds	500.0	10.0	2.0	0.8:1	100	-
Hjordkar	DK	4	Gravel	Reeds	1125.0	175.5	15.6	0.15:1	100	-
Ingstrup	DK	4	Soil	Reeds	115.0	1.2	1.0	0.16:1	100	-
Rugballegard	DK	4	Soil	Reeds	96.0	1.2	1.3	1.5:1	100	-
Knudby	DK	4	Soil	Reeds	361.0	24.2	6.7	1:01	100	-
Borup	DK	4	Soil	Reeds	1440.0	67.7	4.7	1.6:1	100	-
Kalo	DK	4	Soil	Reeds	760.0	269.0	35.4	0.52:1	100	-
Egeskov	DK	4	Soil	Reeds	3600.0	568.8	15.8	-	100	-
Ferring	DK	4	Soil	Reeds	2000.0	58.0	2.9	-	100	-
Fousing	DK	4	Soil	Reeds	1175.0	15.3	1.3	-	100	-
Uggerhalne	DK	4	Soil	Reeds	2640.0	116.2	4.4	-	100	-
Jaungyole	DK	4	Soil	Reeds	800.0	53.6	6.7	-	100	-
Karstoft	DK	4	Soil	Reeds	2400.0	24.0	1.0	-	100	-

LOCATION		REF.	TYPE	VEGETATION	AREA (m^2)	Q (m^3/d)	HLR (cm/d)	L:W Ratio	DEPTH (cm)	ROOT ZONE DEPTH(cm)
Gudum	DK	4	Soil	Reeds	2392.0	162.7	6.8	-	100	-
Sabroe	DK	4	Soil	Reeds	2650.0	225.3	8.5	-	100	-
Ormslev	DK	4	Soil	Reeds	1850.0	129.5	7.0	-	100	-
Fare	DK	4	Soil	Reeds	1500.0	132.0	8.8	-	100	-
Fjaltring	DK	4	Soil	Reeds	2200.0	35.2	1.6	-	100	-
Hammelev	DK	4	Soil	Reeds	1500.0	49.5	3.3	-	100	-
Lyngby	DK	4	Soil	Reeds	3520.0	98.6	2.8	-	100	-
Homa	DK	4	Soil	Reeds	1700.0	96.9	5.7	-	100	-
Borum	DK	4	Soil	Reeds	2500.0	147.5	5.9	-	100	-
Hobjerg	DK	4	Soil	Reeds	4100.0	110.7	2.7	-	100	-
Santee	CA	5	Gravel(P)	Reeds	64.8	3.0	4.7	5.3:1	76	76
Santee	CA	5	Gravel(P)	Bulrushes	64.8	3.0	4.7	5.3:1	76	60
Santee	CA	5	Gravel(P)	Cattails	64.8	3.0	4.7	5.3:1	76	30
Bear Creek	AL	6	Gravel(S)	Cattails	2032.0	57.5	2.8	1.4:1	30	30

References

(1) Watson et al. 1990
(2) Cooper et al. 1990
(3) Cooper et al. 1987
(4) Brix 1987
(4) Schierup and Brix 1990
(4) Brix and Schierup 1989
(5) Gersberg et al. 1986
(6) TVA/WR/WQ-90/55 1990

APPENDIX A-2
SFCW BOD REMOVAL

LOCATION		BOD5in (mg/l)	BOD5eff (mg/l)	LOAD (kg/ha-d)	REMain (kg/ha-d)	% REM
Benton#3	KY	26.0	7.0	15.31	4.12	73.1
Hardin#1	KY	55.0	10.0	39.66	7.21	81.8
Hardin#2	KY	55.0	4.0	39.66	2.88	92.7
Gravesend#1	UK	224.0	70.0	182.86	57.14	68.8
Gravesend#2	UK	224.0	79.0	182.86	64.49	64.7
Gravesend#3	UK	224.0	52.0	182.86	42.45	76.8
Holtby	UK	242.0	44.0	118.63	21.57	81.8
Marnhull#1	UK	93.0	15.0	107.55	17.35	83.9
Marnhull#2	UK	93.0	17.0	107.55	19.66	81.7
Middleton	UK	9.9	2.7	7.70	2.10	72.7
Valley Field	UK	201.0	29.0	759.33	109.56	85.6
Bluther#1	UK	221.0	47.0	234.81	49.94	78.7
Bluther#2	UK	221.0	59.0	234.81	62.69	73.3
Bluther#3	UK	221.0	30.0	234.81	31.88	86.4
Bluther#4	UK	221.0	56.0	234.81	59.50	74.7
Castleroe	UK	80.0	73.0	133.33	121.67	8.8
Moesgard	DK	97.0	23.0	19.40	4.60	76.3
Hjordkar	DK	126.0	35.0	196.56	54.60	72.2
Ingstrup	DK	543.0	42.0	54.30	4.20	92.3
Rugballegard	DK	371.0	37.0	48.23	4.81	90.0
Knudby	DK	96.0	12.0	64.32	8.04	87.5
Borup	DK	83.0	17.0	39.01	7.99	79.5
Kalo	DK	101.0	24.0	357.54	84.96	76.2
Egeskov	DK	20.0	6.0	31.60	9.48	70.0
Ferring	DK	91.0	7.0	26.39	2.03	92.3
Fousing	DK	182.0	25.0	23.66	3.25	86.3
Uggerhalne	DK	108.0	21.0	47.52	9.24	80.6
Jaungyole	DK	59.0	9.0	39.53	6.03	84.7
Karstoft	DK	145.0	27.0	14.50	2.70	81.4
Gudum	DK	76.0	13.0	51.68	8.84	82.9

LOCATION		BOD5in (mg/l)	BOD5eff (mg/l)	LOAD (kg/ha-d)	REMain (kg/ha-d)	% REM
Sabroe	DK	50.0	23.0	42.50	19.55	54.0
Ormslev	DK	43.0	10.0	30.10	7.00	76.7
Fare	DK	26.0	10.0	22.88	8.80	61.5
Fjaltring	DK	223.0	29.0	35.68	4.64	87.0
Hammelev	DK	163.0	107.0	53.79	35.31	34.4
Lyngby	DK	88.0	9.0	24.64	2.52	89.8
Homa	DK	40.0	7.0	22.80	3.99	82.5
Borum	DK	178.0	31.0	105.02	18.29	82.6
Hobjerg	DK	212.0	34.0	57.24	9.18	84.0
Santee	CA	118.3	22.2	55.60	10.43	81.2
Santee	CA	118.3	5.3	55.60	2.49	95.5
Santee	CA	118.3	30.4	55.60	14.29	74.3
Bear Creek	AL	13.0	1.0	3.68	0.28	92.3

APPENDIX A-3
SFCW TN REMOVAL

LOCATION		TNin (mg/l)	TNeff (mg/l)	TN LOAD (kg/ha-d)	TN REMain (kg/ha-d)	% REM
Benton#3	KY	14.3	10.1	8.42	5.95	29.4
Hardin#1	KY	19.5	12.0	14.06	8.65	38.5
Hardin#2	KY	19.5	8.0	14.06	5.77	59.0
Gravesend#1	UK	-	-	-	-	-
Gravesend#2	UK	-	-	-	-	-
Gravesend#3	UK	-	-	-	-	-
Holtby	UK	-	-	-	-	-
Marnhull#1	UK	-	-	-	-	-
Marnhull#2	UK	-	-	-	-	-
Middleton	UK	-	-	-	-	-
Valley Field	UK	-	-	-	-	-
Bluther#1	UK	-	-	-	-	-
Bluther#2	UK	-	-	-	-	-
Bluther#3	UK	-	-	-	-	-
Bluther#4	UK	-	-	-	-	-
Castleroe	UK	-	-	-	-	-
Moesgard	DK	46.0	28.0	9.20	5.60	39.1
Hjordkar	DK	36.0	26.0	56.16	40.56	27.8
Ingstrup	DK	132.0	20.0	13.20	2.00	84.8
Rugballegard	DK	84.0	35.0	10.92	4.55	58.3
Knudby	DK	33.0	19.0	22.11	12.73	42.4
Borup	DK	31.0	20.0	14.57	9.40	35.5
Kalo	DK	46.0	35.0	162.84	123.90	23.9
Egeskov	DK	10.0	7.0	15.80	11.06	30.0
Ferring	DK	19.0	13.0	5.51	3.77	31.6
Fousing	DK	48.0	28.0	6.24	3.64	41.7
Uggerhalne	DK	27.0	21.0	11.88	9.24	22.2
Jaungyole	DK	21.0	12.0	14.07	8.04	42.9
Karstoft	DK	34.0	16.0	3.40	1.60	52.9
Gudum	DK	23.0	15.0	15.64	10.20	34.8

LOCATION		TNin (mg/l)	TNeff (mg/l)	TN LOAD (kg/ha-d)	TN REMain (kg/ha-d)	% REM
Sabroe	DK	34.0	32.0	28.90	27.20	5.9
Ormslev	DK	20.0	15.0	14.00	10.50	25.0
Fare	DK	17.0	13.0	14.96	11.44	23.5
Fjaltring	DK	38.0	23.0	6.08	3.68	39.5
Hammelev	DK	45.0	37.0	14.85	12.21	17.8
Lyngby	DK	29.0	17.0	8.12	4.76	41.4
Homa	DK	26.0	16.0	14.82	9.12	38.5
Borum	DK	53.0	33.0	31.27	19.47	37.7
Hobjerg	DK	47.0	30.0	12.69	8.10	36.2
Santee	CA	27.8	-	13.07	-	-
Santee	CA	27.8	-	13.07	-	-
Santee	CA	27.8	-	13.07	-	-
Bear Creek	AL	48.0	9.0	13.59	2.55	81.3

APPENDIX A-4

SFCW NH_3 REMOVAL

LOCATION		NH_3in (mg/l)	NH_3eff (mg/l)	NH_3 Load (kg/ha-d)	NH_3REMain (kg/ha-d)	%REM
Benton#3	KY	4.8	8.6	2.83	5.07	-79.2
Hardin#1	KY	9.4	10.9	6.78	7.86	-16.0
Hardin#2	KY	9.4	6.5	6.78	4.69	30.9
Gravesend#1	UK	40.0	33.0	32.65	26.94	17.5
Gravesend#2	UK	40.0	33.0	32.65	26.94	17.5
Gravesend#3	UK	40.0	33.0	32.65	26.94	17.5
Holtby	UK	39.0	34.0	19.12	16.67	12.8
Marnhull#1	UK	28.0	10.0	32.38	11.56	64.3
Marnhull#2	UK	28.0	9.0	32.38	10.41	67.9
Middleton	UK	2.3	0.8	1.79	0.62	65.2
Valley Field	UK	26.0	24.0	98.22	90.67	7.7
Bluther#1	UK	-	-	-	-	-
Bluther#2	UK	-	-	-	-	-
Bluther#3	UK	-	-	-	-	-
Bluther#4	UK	-	-	-	-	-
Castleroe	UK	-	-	-	-	-
Moesgard	DK	-	-	-	-	-
Hjordkar	DK	-	-	-	-	-
Ingstrup	DK	-	-	-	-	-
Rugballegard	DK	-	-	-	-	-
Knudby	DK	-	-	-	-	-
Borup	DK	-	-	-	-	-
Kalo	DK	-	-	-	-	-
Egeskov	DK	-	-	-	-	-
Ferring	DK	-	-	-	-	-
Fousing	DK	-	-	-	-	-
Uggerhalne	DK	-	-	-	-	-
Jaungyole	DK	-	-	-	-	-
Karstoft	DK	-	-	-	-	-

LOCATION		NH_3in (mg/l)	NH_3eff (mg/l)	NH_3 Load (kg/ha-d)	NH_3REMain (kg/ha-d)	%REM
Gudum	DK	-	-	-	-	-
Sabroe	DK	-	-	-	-	-
Ormslev	DK	-	-	-	-	-
Fare	DK	-	-	-	-	-
Fjaltring	DK	-	-	-	-	-
Hammelev	DK	-	-	-	-	-
Lyngby	DK	-	-	-	-	-
Homa	DK	-	-	-	-	-
Borum	DK	-	-	-	-	-
Hobjerg	DK	-	-	-	-	-
Santee	CA	24.7	5.3	11.61	2.49	78.5
Santee	CA	24.7	1.4	11.61	0.66	94.3
Santee	CA	24.7	17.7	11.61	8.32	28.3
Bear Creek	AL	10.7	1.8	3.03	0.51	83.2

REFERENCES

Armstrong, W. 1967. Oxygen Diffusion From the Roots of some British Bog Plants. Nature 204:801-802.

Armstrong, W. 1972. A Re-examination of the Functional Significance of Aerenchyma. Phsiol. 27:173-177.

Armstrong, W. 1978. Root Aeration in the Wetland Condition. In: Plant Life in Anaerobic Environments. Hooks, D.D. and Crawford, R.M.M. (eds). Ann Arbor Science Pub., Ann Arbor Mich.

Armstrong, J. and Armstrong, W. 1988. Phragmites Australis - A Preliminary Study of Soil Oxidizing sites and Internal Gas Transport Pathways. New Phytol. 108:373-382.

Armstrong, J and Armstrong, W. 1990. Light Enhanced Convective Throughflow increases Oxygenation in Rhizomes and Rhizospheres of Phragmites Australis. New Phytol. 114:121-128.

Armstrong, J. and Armstrong, W. 1991. A Convective Throughflow of Gases in Phragmites Australis. Aqu. Bot. 39:75-88.

Armstrong, W., Armstrong, J. and Beckett, P.M. 1990. Measurement and Modelling of Oxygen Release From Roots of Phragmites Australis. In: Constructed Wetlands In Water Pollution Control. Cooper, P.F. and Findlater, B.C. (eds). Pergamon Press Inc. Elmsford, New York 10523.

Bacha, R.E. and Hossner, L.R. 1977. Characteristics of Coatings Formed on Rice Roots as Affected by Iron and Manganese Additions. Soil Sci. Am. J. 41:931-935.

Boon, A.G. 1985. Report of a visit by Members and Staff of WRC to Germany to Investigate the Root Zone Method for Treatment of Wastewaters. Water Research Center. Stevenage, England.

Bowmer, K.H. 1987. Nutrient Removal From Effluents By An Artificial Wetland: Influence Of Rhizosphere Aeration And Preferential Flow Studied Using Bromide And Dye Tracers. Wat. Res. 21:591-599.

Breen, P.F. 1990. A Mass Balance Method For Assessing The Potential Of Artificial Wetlands For Wastewater Treatment. Wat. Res. 24:689-697.

Brix, H. 1987. Treatment of Wastewater in the Rhizosphere of Wetland Plants - The Root Zone Method. Wat. Sci. Tech. 19:107-118.

Brix, H. 1988. Light Dependent Variations in the Composition of the Internal Atmosphere of Phragmites Australis. Aquat. Bot. 30:319-329.

Brix, H. 1989. Gas Exchange Through Dead Culms of Reeds, Phragmites Australis. Aquat. Bot. 35:81-98.

Brix, H. and Schierup, H.H. 1989. Sewage Treatment in Constructed Reed Beds - Danish Experiences. Wat. Sci. Tech. 21:1665-1668.

Brix, H. and Schierup, H.H. 1990. Soil Oxygenation in Constructed Reed Beds: The Role of Macrophyte and Soil-Atmosphere Interface Oxygen Transport. In: Constructed Wetlands In Water Pollution Control. Cooper, P.F. and Findlater, B.C. (eds). Pergamon Press Inc. Elmsford, New York 10523.

Brock, T.D. and Madigan, M.T. 1991. Biology of Microorganisms. Prentice Hall, Englewood Cliffs, New Jersey 07632.

Bucksteeg, K. 1985. Initial experience of the Root Zone Process in Bavaria. In: Proc. Symposium held at the Technical University of Darmstadt, July 11 1985.

Burgoon, P.S., Debusk, T.A., Reddy, K.R., and Koopman, B. 1991. Vegetated Submerged Beds With Artificial Substrates, I: BOD Removal & II: N and P Removal. Jou. Env. Eng. 17:394-424.

Chalk, E. and Whealeon, G. 1989. The Root-Zone Process at Holtby Sewage Treatment Works. J. IWEM. 3:201-207.

Choate, K.D. 1989. First Semiannual Monitoring Report: Demonstrations of Constructed Wetlands for Municipal Wastewater, March to December 1988. TVA/WR/WQ--89/5. Chattanooga, Tennessee.

Conley, L.M., Dick, R.I., and Lion, L.W. 1991. An Assessment of the Root Zone Method of Wastewater Treatment. Res. J. WPCF. 63:239-247.

Cooper, P.F. and Boon, A.G. 1987. The Use of Phragmites for Wastewater Treatment by The Root Zone Method: The U.K. Approach. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Davis, D.G. and Montgomery, J.C. 1987. EPA's Regulatory And Policy Considerations On Wetlands And Municipal Wastewater Treatment. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Dodd, R.C., Davis, J.A., and Kerr, M. 1986. The use of wetlands for wastewater management: An Update. Presented at 1986 Triangle Conference on Environmental Technology, University of North Carolina, Chapel Hill.

EC/EWPCA. 1990. European Design and Operations Guidelines for Reed Bed Treatment Systems. Cooper, P.F. (eds). WRc Swindon, Swindon England.

Environmental Protection Agency (EPA). 1981. Process Design Manual For Land Treatment Of Municipal Wastewater. EPA/625/1-81-013. U.S. Environmental Protection Agency, Office of Research and Development, Center for Environmental Research Information, Cincinnati, OH 45268.

Environmental Protection Agency (EPA). 1985. Freshwater wetlands for wastewater management handbook. EPA 904/9-85-135. Region IV, Atlanta GA.

Environmental Protection Agency (EPA). 1988. Design Manual: Constructed Wetlands and Aquatic Plant Systems for Municipal Wastewater Treatment. EPA/625/1-88/022. U.S. Environmental Protection Agency, Office of Research and Development, Center for Environmental Research Information, Cincinnati, OH 45268.

Findlater, B.C., Hobson, J.A., and Cooper, P.F. 1990. Reed Bed Treatment Systems: Performance Evaluation. In: Constructed Wetlands In Water Pollution Control. Cooper, P.F. and Findlater, B.C. (eds). Pergamon Press Inc. Elmsford, New York 10523.

Gambrell, R.P. and Patrick, W.H. 1978. Chemical and Microbiological Properties of Anaerobic Soils and Sediments. In: Plant Life in Aerobic Environments. Hook, D.D. and Crawford , R.M.M. (eds). Ann Arbor Science, Ann Arbor, MI.

Gearheart, R.A., Finney, B.A., Wilbur, S., Williams, J., and Hull, D. 1985. The Use of Wetland Treatment Processes in Water Reuse. In: Future of Water Reuse,V2, AWWA Research Foundation. Denver, CO.

Gersberg, R.M., Elkins, B.V., and Goldman, C.R. 1983. Nitrogen Removal In Artificial Wetlands. Water Res. 17:1009-1014.

Gersberg, R.M., Elkins, B.V., and Goldman, C.R. 1984. Use of Artificial Wetlands to Remove Nitrogen from Wastewater. Jou. WPCF. 56:152-156.

Gersberg, R.M., Elkins, B.V., Lyon S.R., and Goldman C.R. 1986. Role of Aquatic Plants in Wastewater Treatment by Artificial Wetlands. Wat. Res. 20:363-368.

Good, B.J. and Patrick, W.H. 1987. Root-Water-Sediment Interface Processes. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Hammer, D.A. and Bastian, R.K. 1987. Wetlands Ecosystems: Natural Water Purifiers? In: Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural. Hammer, D.A. (eds). Lewis Pub. Inc.

Herskowitz, J., Black, S., and Lewandowski W. 1987. Listowel Artificial Marsh Treatment Project. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Hyde, H.C. and Ross, R.S. 1984. Technology Assessment of Wetlands for Municipal Wastewater Treatment. EPA/600/2-84/154. U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory, Office of Research and Development, Center for Environmental Research Information, Cincinnati, OH 45268.

Jones, A.A. 1992. Microbial Rock Plant/Microbial Rock Filters, Emerging Low Cost Low O&M Technologies. Presented at The First Symposium On Site Wastewater Treatment and Research, Austin TX, Aug. 9-11.

Kadlec, R.H. and Kadlec J.A. 1979. Wetlands and Water Quality. In: Wetland Functions and Values: The State of Our Understanding. Greeson, P.E., Clark, J.R., and Clark, J.E. (eds). American Water Resources Association, Minneapolis, MN.

Kadlec, R.H. and Tilton, D.J. 1979. The use of freshwater wetlands as a tertiary wastewater treatment alternative. CRC Critical Reviews in Environmental Control. 9:185-212.

Lawson, G.J. 1985. Cultivating Reeds for Root Zone Treatment of Sewage. Report 965. Institute of Terrestrial Ecology. Cambridge, England.

Nichols, D.S. 1983. Capacity of Natural Wetlands to Remove Nutrients from Wastewater. Jou. WPCF. 55:495-505.

Nixon, S.W. and Lee, V. 1986. Wetlands and water quality: A regional view of recent research in the United States on the role of freshwater and saltwater wetlands as sources, sinks, and transformers of nitrogen, phosphorus, and various heavy metals. Technical Report Y-86-2. U.S. Army Corps of Engineers, Vicksburg, Mississippi.

Ponnamperuma, R.N. 1965. Dynamic Aspects of Flooded Soils and the Nutrition of the Rice Plant. In: The Mineral Nutrition of the Rice Plant. John Hopkins Press, Baltimore MD.

Reddy, K.R., D'Angelo E.M., and Debusk T.A. 1989. Oxygen Transport Through Aquatic Macrophytes: The Role in Wastewater Treatment. J. Environ. Qual. 19:261-267.

Reed, S.C. 1991. Constructed Wetlands For Wastewater Treatment. Biocycle. Jan:44-49.

Reed, S.C. and Bastian, R. 1985. Wetlands for Wastewater Treatment: An Engineering Perspective. In: Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold, NY.

Reed, S.C., Bastian, R., and Jewell, W. 1979. Engineering Assessment of Aquaculture Systems for Wastewater Treatment: An Overview. In: Aquaculture Systems for Wastewater Treatment: Seminar Proceedings and Engineering Assessment. EPA 430/9-80/006. U.S. Environmental Protection Agency.

Reed, S.C. and Brown, D.S. 1992. Constructed wetland design - the first generation. Water Environ. Res. 64:776-781.

Reed, S.C., Middlebrooks, E.J., and Crites, R.W. 1988. Natural Systems for Waste Management & Treatment. Mcgraw Hill Pub. Inc.

Reed, S.C., Middlebrooks, E.J., and Crites, R.W. 1992. Natural Systems for Waste Management & Treatment, 2nd. Edition. Mcgraw Hill Pub. Inc., (in press).

Richardson, C.J. and Davis, D.G. 1987. Natural And Artificial Wetland Ecosystems: Ecological Opportunities And Limitations. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Richardson, C.J. and Nichols, D.S. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. In: Ecological Considerations in Wetlands Treatment of Municipal Wastewater. Godfrey, E.R., Keynor, E.R., Pelczarski, S., and Benforedo, J. (eds). Van Nostrand Reinhold Co., New York.

Rogers, K.H., Breen P.F., and Chick A.J. 1991. Nitrogen Removal in Experimental Wetland Treatment Systems: Evidence for the Role of Aquatic Plants. Res. Jou. WPCF. 63:934-941.

Shearer, L., Goldsmith, J., Young, C., Kearns, O., and Tamplin, B. 1972. Methemoglolbin levels in infants in an area with high nitrate water supply. Am. J. Publ. Hlth. 62:1174-1180.

Schierup, H.H. and Brix, H. 1990. Danish Experience With Emergent Hydrophyte Treatment Systems (EHTS) and Prospects in the Light of Future Requirements on Outlet Water Quality. Wat. Sci. Tech. 22:65-72.

Seidel, K. 1976. Macrophytes and Water Purification. In: Biological Control of Water Pollution. Tourbier J. and Pierson R.W. (eds). University of Pennsylvania Press, Philadelphia, PA.

Smith, A.I. 1987. Wastewater: A Perspective. In: Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural. Hammer, D.A. (eds). Lewis Pub. Inc.

Stanford, G. 1975. Effect of Temperature on Denitrification Rate in Soils. Soil Sci. Soc. Am. Proc. 39:867.

Stowell, R., Ludwig, R., Colt, J., and Tchobanoglous G. 1981. Concepts in aquatic treatment systems design. J. Env. Eng. Div., Processings of the ASCE 107, No. EE5.

Tchobanoglous, G. and Burton, F.L. 1991. Wastewater Engineering: Treatment, Disposal, and Reuse. Mcgraw Hill Pub. Inc.

Tchobanoglous, G. and Culp, G.L. 1980. Wetland systems for wastewater treatment. In: Aquaculture Systems for Wastewater Treatment: An Engineering Assessment. Reed S.C. and Bastian R.K. (eds). EPA 430/9-80-007. Washington DC.

Water Pollution Control Federation (WPCF). 1990. Natural Systems for Wastewater Treatment, Manual of Practice. Reed S.C. Water Pollution Control Federation, Alexandria VA.

Watson J.T. 1992. Constructed Wetlands For Municipal Wastewater Treatment: State Of The Art. TVA report presented at the symposium Epuration Des Eaux Usees Par Les Plants: Perspectives D'Avenir Au Quebec. Montreal, Quebec, March 20, 1992.

Watson, J.T., Reed, S.C., Kadlec, R., Knight, R.L., and Whitehouse A.E. 1988. Performance Expectations and Loading Rates for Constructed Wetlands. Prepared for The International Conference on Constructed Wetlands for Wastewater Treatment. Chattanooga,Tennessee, June 13-17.

Wolverton, B.C. 1987. Aquatic Plants For Wastewater Treatment: An Overview. In: Aquatic Plants for Water Treatment and Resource Recovery. Reddy, K.R. and Smith, W.H.(eds). Magnolia Publishing Inc., Orlando, Fl.

Wolverton, B.C. 1988. Aquatic Plants Wastewater Treatment System. Paper presented at the National Environmental Health Association Conference. Mobile, Alabama, Feb.7-9.

Wolverton, B.C. 1980. Higher plants for recycling human waste into food, potable water, and revitalized air in a closed life support system. NASA/ERL Report No. 192.

Wolverton, B.C. and Mcdonald, R.C. 1976. Don't Waste Waterweeds. New Scientist. 71:318-320.

Wolverton, B.C., Mcdonald, R.C., and Duffer, W.R. 1983. Microorganisms and Higher Plants for Wastewater Treatment. J. Environmental Quality. 12:236-242.

Zachritz, W.H. and Fuller, J.W. Unpublished Article. Performance of an Artificial Wetlands Filter Treating Facultative Lagoon Effluent At Carville, LA. U.S. Public Health Service, Gillis W. Long Hansen's Disease Center, Carville, LA 70721.

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face flow constructed wet-
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